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Measuring and Comparing the Cost-Effectiveness of EPA Regulatory Efforts to Control Toxics-Related Health Risks
Volume I: Feasibility Study

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MEASURING AND COMPARING THE COST-EFFECTIVENESS OF EPA REGULATORY EFFORTS TO CONTROL TOXICS-RELATED HEALTH RISKS--VOLUME I, FEASIBILITY STUDY

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FOREWORD

The Office of Toxics Integration (OTI) began this study of cost-effectiveness analysis with several objectives in mind. First, it was important to determine the extent to which traditional cost-effectiveness methods might be used to contribute to OTI's inter-media integration responsibilities for chemicals. Second, in view of the rather unique character of an integrated or multi-media regulatory analysis, some of the conceptual as well as practical limitations of most cost-effectiveness analytical practices available to the Agency were already evident. Thus, this study was designed to explore ways to bridge those limitations, to develop several innovative techniques and concepts, and (in cases where the limitations may be problematic) to get a better understanding of the implications for multi-media analysis.

OTI's mission is to integrate the EPA program offices' activities with respect to toxic chemicals of multi-media interest. A major part of that responsibility is to develop integrated, Agency-wide strategies for toxic substances problems occurring in more than one medium. In general, the primary objectives of an integrated EPA control strategy are to minimize inadvertent transfers of a toxics chemical problem from one medium to another and to maximize the combined (i.e., all media taken together) regulatory benefits and minimize the combined regulatory costs. Because an integrated strategy specifically considers all media and regulatory options, controls can be designed to take advantage of variances across media in exposure levels, costs of control, etc. At the same time, an integrated approach allows OTI to determine the net effects of all controls and to evaluate the implications of transfers of toxics from one medium to another.

When viewed as a whole, a set of regulatory proposals representing each EPA program office's effort to control a substance in each medium reflects certain objectives regarding benefits and costs. OTI's goal in developing inter-media cost-effectiveness analysis is to allow comparative evaluation of combined alternatives for chemical risk reduction. The key analytical problem in developing an integrated approach, however, is the difficulty of comparing the relative costs and "benefits" of regulatory proposals (or existing regulations) across media.

Cost-effectiveness analysis is traditionally viewed as an appropriate analytical technique for identifying least-cost ways of achieving a given objective, or for maximizing the benefits for a given cost. However, most examples of cost-effectiveness analysis in the Agency have been used in a single medium context (e.g., cost-effectiveness comparisons of alternative control technologies in air). Because these analyses are somewhat narrow in focus and single-media in context, conventional benefit measures such as "pounds of pollutant emissions controlled" may be adequate for single-media analytical requirements.

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The Office of Toxics Integration of the Environmental Protection Agency undertook this study to examine the conceptual and practical limitations of the application of traditional cost-effectiveness methods to multi-media regulatory analysis and to determine the extent to which these methods may contribute to the integration of EPA's multiple program office controls on toxic chemicals.

The goal in developing inter-media air, land, and water cost-effectiveness analysis is to allow comparative evaluation of the various combined alternatives for chemical risk reduction. The key analytical problem in developing an integrated approach is the difficulty of comparing the relative costs and benefits of regulatory proposals (or existing regulations) across media. Conventional benefit measures are usually not feasible to support a multi-media chemical control strategy. Benefit measures must reflect some positive change that is closer in the chain of effects to the ultimate receptors, e.g., units of exposure reduced or specific health risks reduced, than has generally been reflected in past regulations. The study also demonstrates the difficulties in comparing cost data between different Agency control programs.

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However, because of the broad scope and objectives of an inter-media analysis, conventional benefit measures are usually not sufficient for inter-media cost-effectiveness analysis. To the extent that OTI's cost-effectiveness analyses support a multi-media control stategy, benefit measures must reflect some positive change that is closer in the chain of effects to the ultimate receptors. By considering and using measures such as units of exposure reduced, or even specific health risks reduced, an integrated control strategy based on an inter-media cost-effectiveness analysis can take advantage of the likely variation across media in the cost of control per exposure reduced, health risk reduced, etc.

At the same time, this study demonstrates the difficulties in comparing cost data between different control programs. For example, inconsistencies arising out of differences in cost models used, or simply the ways in which costs are categorized, present analytical and practical problems for a multi-media cost-effectiveness analysis. This study highlights some of these problems and identifies some candidate solutions for them.

The first part of this study is a discussion of some of the major analytic principles that are the basis for the subsequent analyses. This part also contains a discussion of the major conceptual and practical problems in applying these concepts.

The remainder of the document, including the second volume, comprises a series of case studies. In Volume I, these are less detailed, pilot case studies. Volume II contains a more in-depth approach designed to add substance to some of the concepts and principles discussed earlier.

The regulations studied in the course of this work were chosen primarily to highlight the analytical issues, but also to begin collection of baseline data which will provide a frame of reference for future analyses.

Walter W. Kovalick Jr

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1. INTRODUCTION AND SUMMARY

Because resources available to control toxic hazards are limited, it is appropriate for cost-effectiveness to be considered when deciding such issues as which toxic hazards will receive priority attention, in what media, and using which legislative mandate. The Toxic Substances Control Act (TSCA) specifically requires cost-effectiveness considerations:

- Section 6(c) requires EPA, when acting under TSCA regulatory authority, to consider the effectiveness and costs of other relevant non-TSCA controls.
- Section 9 requires EPA to insure that the most cost-effective measures are used to control chemical risks that are candidates for TSCA control.

In addition, Executive Order 12291 which deals with federal regulation requires that costs and effectiveness of regulations be considered in setting regulatory priorities and in choosing regulatory alternatives.[1]

The term "cost-effectiveness analysis" strictly speaking, has been applied to the methods used in comparing policy alternatives in which either a clearly defined objective has been specified and the alternative programs under study differ only in cost, or more rarely, in situations where the costs of programs are similar, but the alternative programs differ in the extent of benefits they produce.[2] Neither of these conditions holds true for many of the regulations promulgated and enforced by EPA.

For this reason, we have chosen a broader, more useful interpretation of the term "cost-effectiveness analysis." In this study we take "cost-effectiveness analysis" to mean any analysis that attempts to compare quantitative estimates of the positive and negative outcomes (socially desirable outcomes and social costs) of regulatory programs, in their natural units. That is, no attempt is made to monetize all of the various outcomes of regulatory actions where such monetization is not firmly grounded, or where existing models and techniques do not allow monetization to be performed. For example, rather than attempt to develop monetary equivalents for the poorly defined health consequences of lead in air and in drinking water, we instead develop a framework within which these different kinds of outcomes can be compared, within regulations and between regulations.

Throughout this report, we use the terms "effectiveness" and "benefits" somewhat interchangeably. We view them as equivalent terms in the context of this report. "Effectiveness" refers to some quantity of a desired outcome, such as cancer deaths prevented or life years saved. Many times "benefits" imply the monetization of a quantity of a desired outcome; however this is not a necessary condition for using the term because benefits can also be measured in natural units such as cancer deaths prevented or life years saved.

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[1] "Federal Regulation," Executive Order 12291, February 17, 1981.

[2] E. Stokey, and R. Zeckhauser, A Primer for Policy Analysis (New York, New York: Norton, 1978), pp. 153-154.

The methods we develop could be used to help EPA in evaluating and comparing regulations and in complying with the various requirements for regulatory analysis set forth in TSCA and in Executive Order 12291. Specifically, the methods we suggest could be used to:

- help EPA select effectiveness goals and evaluate costs so that cost-effectiveness analysis, narrowly defined, can be used in comparing alternative regulatory options;
- help EPA conduct analysis aimed at assuring the most efficient allocation of regulatory resources, that is, to help EPA maximize the net benefit of its regulations for a given level of resource expenditure;
- help EPA select priorities for future regulatory attention and existing standards for possible revisions; and
- compare the costs and effectiveness of regulatory alternatives for given substances under different statutes, and for different substances under the same statute.

In addition, the process of developing these methods will:

- develop generic methodologies for performing cost-effectiveness analysis;
- point out the need for data not now routinely developed; and
- point out opportunities for more effective integration of effort across EPA programs.

Before work is begun to develop the cost-effectiveness numbers for existing EPA toxics regulations, it is most important to determine the feasibility of that effort. For the effort to be feasible, three conditions must be met:

1. data needed to perform the analysis must be either available or possible to develop,
2. potential methodological difficulties must be resolvable, and

3. the work must be able to be accomplished within reasonable budgetary constraints.

It is to the question of feasibility that the first volume of this study is addressed.

This feasibility analysis began with a review of EPA toxics regulations promulgated under the following statutes:

- Clean Air Act
- Federal Water Pollution Control Act
- Toxic Substances Control Act
- Safe Drinking Water Act
- Resource Conservation and Recovery Act
- Federal Insecticide, Fungicide, and Rodenticide Act

A main purpose of that review was to develop a list of issues potentially confounding the successful accomplishment of the cost-effectiveness analysis. Refinements to the list were made as the case studies, discussed below, were conducted. The analysis of the issues constitutes a major part of a feasibility analysis.

Chapters 2 and 3 present the analysis of the benefit and cost issues, respectively. The issue discussions include many references to specific EPA toxics regulations. This is because, although a particular issue might theoretically represent a significant obstacle to accomplishing the purposes of the cost-effectiveness analysis, in actuality it might occur so infrequently or be of such small magnitude to not require further attention. Where significant obstacles are found, solutions are discussed. No final solution is proposed for several issues. For those issues, solutions could be developed as required in the actual cost-effectiveness analysis. We did not find any of the issues to represent absolute barriers to performing a cost-effectiveness study.

Chapter 4 presents an introduction to the case studies (cost-effectiveness analyses) performed for the following five EPA toxics regulations:

1. Lead in ambient air (Clean Air Act, Section 109)
2. Lead (and lead compounds) spilled into navigable waters (Federal Water Pollution Control Act, Section 311)
3. Lead in drinking water (Safe Drinking Water Act)
4. Vinyl Chloride emissions (Clean Air Act, Section 112)

5. Chlorofluorocarbons used as aerosol propellants (Toxic Substances Control Act).

The actual case studies are in the appendices to this report.

The purposes of the case studies were to provide an opportunity to further study the benefit and cost issues and to develop a methodology for further cost-effectiveness analyses. The five case studies illustrate well the differences between regulations in terms of analytical treatment required to study them on a cost-effectiveness basis. Also well illustrated are some of the limitations involved in making cost-effectiveness comparisons between regulations.

Chapter 5 discusses the overall conclusions of the feasibility study. In general, we believe a cost-effectiveness analysis of EPA regulations is feasible and can provide useful information. First, needed data exist or can be developed for a sufficient number of toxics regulations to be studied. Second, there do not appear to be any methodological difficulties of sufficient magnitude to jeopardize performance of the study. Third, it appears that the study can be performed within reasonable budgetary constraints.

This feasibility analysis was performed over a relatively short period of time. Because of this, it was not possible to fully review all EPA toxics regulations nor was it possible to fully develop a methodology for performing cost-effectiveness analyses. For purposes of determining feasibility, we believe this review has been sufficient. Nevertheless, one must be cautious in trying to use the information developed in this study beyond the study's purposes.

Volume II of this study contains detailed case studies of the application of cost-effectiveness analysis to EPA toxics regulations. Specific chemicals for study were selected from a series of chemicals identified by EPA as Intermedia Priority Pollutants. The pollutants represent a consensus view of the various EPA programs involved in developing or implementing toxic pollutant regulations. Intermedia Priority Pollutants are principally of concern in more than one media and represent current environmental problems. From the list of Intermedia Priority Pollutants, the following regulations or proposed regulations were selected:

- Lead
    - Clean Air Act, Lead-in-Fuel Regulations
- Asbestos
    - Clean Air Act, National Emission Standard For Hazardous Air Pollutants
    - Clean Water Act, Effluent Limitation Guidelines for Asbestos Manufacturing Point Sources

- Benzene
  - Clean Air Act, National Emission Standard for Hazardous Air Pollutants; Maleic Anhydride Plants, Ethylbenzene/Styrene Plants, and Benzene Storage Vessels.

The detailed case studies pointed out that the conceptually ideal and "correct" cost-effectiveness analysis may often be frustrated by a number of limitations. For example, in the benzene case we found the range of uncertainty underlying the leukemia incidence estimates to be so large such that comparisons of the benzene regulations on the basis of the cost per amount of reduced leukemia incidence may not be very meaningful. Similarly confounding problems arose in the lead and asbestos studies. Thus the analyst's ability to provide definitive cost-effectiveness results is limited.

At a minimum, a measure of the amount of the pollutant that is controlled can usually be reliably quantified. This can be useful information; however results based on this measure of effectiveness must be used only with caution and an awareness of the remaining elements which are necessary to correlate changes in the amount of pollutant controlled with measurable changes in health effects.

2. BENEFIT ISSUES

2.1 INTRODUCTION

In order to analyze the cost-effectiveness of EPA regulations, it is necessary to have a uniform measure of benefits, or a way to value the different type of benefits so they can be translated into comparable units. This chapter discusses several issues regarding various ways of describing and measuring benefits.

The first section describes the components of the ideal benefits analysis, which allows us to highlight specific points where regulatory action has an effect. The ideal analysis and our review of EPA toxics regulations indicates several issues which must be addressed in evaluating the benefits of a regulation:

1. Availability of Benefits Data
2. Quality of the Benefits Data
3. Treatment of Joint Benefits
4. Technological Innovation Due to New Regulations
5. Valuation of the Benefits of Regulations
6. Comparing Costs and Benefits in Different Time Periods.

This chapter focuses on these issues. We describe each issue and point out where it appears to be a problem in EPA regulations. Where an issue poses a problem to the evaluation of a regulation's benefits, we discuss alternative approaches and solutions.

The emphasis of our task was to measure the human health benefits which accrue from the regulations. While we have focused specifically on the benefits of regulations with respect to human health, we recognize the importance of all aspects of the general environment to health and welfare. Where general environmental benefits are a significant part of the impact of a regulation, consideration of only the human health benefits will understate the total benefits.

2.2 THE IDEAL BENEFITS ANALYSIS

2.2.1 Introduction

The ideal benefits analysis can be thought of as a process which describes all of the changes that occur in the environment as a result of regulatory intervention. Exhibit 2-1 presents a model which traces the actual physical processes by which pollutants are emitted, transmitted and affect the environment. The model assumes that some economic activity (for example,

EXHIBIT 2-1

ENVIRONMENTAL IMPACT MODEL*

ACTIVITY (level of production, consumption)

↓ emission factor or other model

DISCHARGE (release of pollutant from commercial control)

↓ models of environmental transport, transformation

PRESENCE (pollutant levels in ambient environment)

↓ complex behaviors, other modifying factors

EXPOSURE (pollutant concentration at relevant locations, actual dosage to receptor organisms)

↓ individual, population dose response relationships, physical damage) functions

EFFECT (direct physical effects of exposure) manufacturing) or some consumption activity (such as driving automobiles) results in the discharge of a pollutant into the environment; the pollutant then undergoes transformation and transport through the environment, resulting in some pattern of distribution or presence of the pollutant in the environment. This pattern of distribution then interacts with some pattern of activity (breathing, drinking, eating, swimming) or with other modifying factors to produce an effect on the environment. This effect, usually measured in some kind of natural physical units (cases of cancer, millimeters of corrosion, deaths) can then be valued by whatever economic or social valuation methodology one wishes to apply. A regulation which brings about some changes in the cycle, such as in an activity or discharge, will alter the subsequent processes which can be measured, resulting in some measure of the benefits of the regulation.

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* This analytical structure draws heavily upon discussions with Dr.Dale Hattis and others at the Center for Policy Alternatives at the Massachusetts Institute of Technology.

The farther along through this process that one is able to go, the more useful one's benefit estimate is likely to be in that the results will be in terms of things that society values (lives saved, dollars saved) and can be compared across regulations. In a case where little information is available, one might have to settle for less easily understood benefits, such as reduced emission of pollutants, or reduced ambient pollutant levels.

In selecting regulations for study, one should go step-by-step through the "ideal" benefits model, paying attention to how one gets from each stage of the model to the next. In the next section, we discuss questions to be considered at the different stages of the benefits analysis.

2.2.2 Discussion of Stages in the Environmental Impact Model

A. Activity ——► Discharge

The first issue to be dealt with is developing some idea of the relationship between the activity and pollutant discharge. Do we know how the regulation affected the polluting activity? What were the levels of discharge before and after regulation? It is often necessary to take into account compliance strategies and timetables, control technologies, or changes in output which affect discharge of pollution. In the case of some of the regulations under consideration for the present study, the problem is a little easier because the analyses are retrospective and historical data may be available. The most straightforward situation is in the case of a ban, or regulation so strict that the process producing the pollution is abandoned. Emission-type regulations (such as a standard limiting the amount of vinyl chloride that can be discharged into the atmosphere), are also fairly straightforward if it can be assumed that 100 percent compliance is achieved and if we know what the emission levels were before the regulation. Otherwise, more complex models of the relationship between the regulated activity and pollutant discharge must be developed.

B. Discharge ——► Presence

At this stage, air and water pollution engineers play an important role. There are numerous models that are commonly used to predict concentrations of pollutants in air or water as a function of discharge patterns. For this analysis, it is not appropriate for us to develop any new models, or to use any other than the simplest, most straightforward variety. We might, therefore, look at regulations where the simplest assumptions about environmental transport have been made, or where they can be used as good approximations. For example, simple rollback models may be legitimately used in thinking about pollution reduction from mobile or area sources in urban areas (automobile emission or fuel quality standards, for example). Again, in some cases (ambient air and water pollutants, for example), the standards themselves may relate to presence levels, and we may be able to assume precisely 100 percent compliance.

Complex environmental transport models are likely to be a problem in cases where pollutants are transported through more than one medium (hazardous waste leaching through soil to groundwater) or where pollutants undergo physical or chemical changes in the environment (chlorofluorocarbons in the stratosphere or ozone production from NOx and hydrocarbons). In at least some of these cases (chlorofluorocarbons, for example) detailed calculations have been performed that allow predictions to be made about changes in ambient levels that would be brought about by regulation. An additional problem is that pollutant levels may change slowly over time and detailed prediction about the temporal patterns of ambient concentration need to be developed, if accurate measures of benefits are to be made.

C. Presence ⟶ Exposure

Once an agent is present in the ambient environment, the actual degree of exposure experienced by the receptors (humans, animals, or materials) is often greatly influenced by complex human behaviors. For example, do people stay indoors at home during high ozone hours of the day, do they commute to work through heavily polluted industrial areas, or do they build neighborhoods near chemical dumps? How much air do they breathe during the day; how much water do they drink? All of these factors may greatly influence the amount of pollutants to which they are exposed.

Again, for the purposes of this analysis, we often need to make simplifying assumptions about these complex behavior patterns. Some can be treated as background variables that do not differ between the regulated and unregulated situations. In other instances, the existing studies have made some kinds of simple assumptions which can be accepted in lieu of further data, such as the amount of air that an average person breathes or the amount of water an average person drinks.

D. Exposure ⟶ Effect

Much research has been devoted to the problem of relating time-exposure patterns of pollutants in the environment to the degree of damage produced. The most thoroughly studied impacts have been those dealing with acute toxic and carcinogenic effects on humans, acute air pollutant episodes producing damage to commercial crops, and the effect of corrosive air pollutants on building materials. There are some data available as to the relationship between pollutant exposure and damage to individual wild plant and animal species. Analysis of other types of pollutant-induced damage (for example, chronic toxic effects on humans or pollutant effects on complex ecosystems) is more difficult.

It is generally more feasible (although not necessarily straightforward or non-controversial) to analyze the following: (1) carcinogenic effects on humans, where animal data or good epidemiological studies are available, (2) acute and classic chronic toxic effects on humans, where good data exists from case studies of exposed individuals, (3) damages to commercial crops from ozone or other airborne pollutant exposure, and (4) air pollution damage to building materials from SO_2 and other pollutants.

More difficult analyses are: (1) predicting carcinogenic effects on humans where no animal data or epidemiological studies are available, (2) measuring chronic functional impairments (decreases in lung, kidney function, for example) produced by chronic pollutant exposure, (3) quantifying the effects of pollutant exposure on individual wildlife species. In the case of chronic functional impairments, one may actually have two tasks: relating exposure patterns to changes in function and then relating changes in function to final health outcomes, such as morbidity and mortality.

The most difficult analyses are: (1) measuring mutagenic or behavioral effects on humans in any quantitative way, and (2) measuring the long term effects of pollutant exposure on complex ecosystems. If most of the pollutant's effects fall in these categories, then quantitative analysis is most likely infeasible.

2.2.3 Summary

In the ideal analysis, we would like to develop two very clear pictures; one a picture of the temporal and spatial patterns of activity, emission, presence, exposure, and effect that would have occurred if the regulation in question had not been promulgated. The other picture would be similar, except that it describes exactly what has occurred and will occur as a result of the regulation. Each step in the analysis would be linked by precisely defined models relating presence to exposure, exposure to effect, and so on. In a full scale benefits analysis, we would expect to explore each step of this environmental impact model.

In this analysis, we will have to settle for relatively simple assumptions about what is likely to have happened in the absence of regulation and rely on simple approximations for many of the models in the analysis (rollback models, linear dose-response, etc.). To the extent that adequate modeling has been performed previously, the development of benefits estimates will be greatly facilitated.

We have emphasized the use of exposure data and the development of dose response data in order to measure effects for each regulation. A large part of the discussion of the benefits concentrates on the assumptions involved in using this kind of data. There is no easy answer to estimating the degree of uncertainty surrounding benefit estimates. In this study, we will make our best estimates of uncertainty, even if they are subjective, and we will not discard any information at any stage of the analysis. Any benefits estimate we develop is likely to be highly uncertain; we believe that it is better to say so than to present unrealistically precise estimates.

2.3 DISCUSSION OF ISSUES

In this section, we discuss issues identified as important in determining the feasibility of benefits analysis of EPA toxics regulations.

2.3.1 Availability of Benefits Data

The comparison of benefits in terms of health effects of the different regulations depends on the availability of specific benefits estimates or the availability of the data to make these estimates. In other words, it is necessary to know the actual changes in health which are brought about as a result of each particular regulation. In many cases, EPA predicts the effect of a regulation in its support documents, but does not translate the data into a particular statement of risk reduction anticipated or accomplished. The statutes which mandate EPA action do not specify that risk reduction estimates be made. Although there seems to be increasingly more thorough risk assessments with more recent regulations, in only a few has the actual process of risk reduction analysis been performed. We discuss the use of the data in the risk assessments in the next section; in this section we review the different EPA toxics regulations to demonstrate where risk reduction estimates have been made.

The Safe Drinking Water Act directs EPA to take action on contaminants which "may have an adverse effect on health." Thus, the goal is set to achieve a particular standard of healthiness (i.e., no adverse health effects from drinking water), and without measuring how far it has to go to get there. Even though the control options have to be "feasible", EPA has not looked at the values of different levels of health to balance the projected costs. Some regulations lack a specific measurement of benefits at all. The first group of Interim Primary Drinking Water Regulations published under the Safe Drinking Water Act are examples of the emphasis in collecting data only about the toxicity of a substance. In setting the maximum contaminant levels (MCLs) for sixteen organic and inorganic chemicals, EPA demonstrated that each of the contaminants is highly toxic. In some cases, there is evidence that the risk is linked to their presence in drinking water (skin cancer is correlated with high concentrations of arsenic in drinking water, for instance), while in others, there is no proven direct link, but only evidence that the contaminant is toxic and that it is present in drinking water. EPA did demonstrate that reduction in the levels of the contaminants in drinking water would decrease the total human exposure of each, by calculating the daily intake of the average male. Implicit in these standards is the assumption that lowering the contaminant level would improve health, but there is no specifically demonstrated level of improved health that would be achieved with each standard. Also, EPA did not calculate what levels were currently in the drinking water systems in the country, and what overall levels would be achieved with the new regulations. Thus, much work is involved in gathering the data to predict the health benefits for any one of the MCLs established, or to predict the benefits of the whole set of MCLs. For this reason, the Safe Drinking Act Water Standard for lead was included as a case study (Appendix A).

The amendments to the Interim Primary Drinking Water Regulations did project the effects of the proposed MCLs. Thus, for trihalomethanes EPA used the dose-response data on the effects of animal ingestion of chloroform (one of the trihalomethanes included in the standard) to calculate that the proposed standard would decrease the number of cancers in the affected population by 200 to 500 cases over 70 years of exposure.

When EPA published the regulation to control polychlorinated biphenyls as mandated by the Toxic Substances Control Act, it demonstrated a wide range of severe adverse effects of PCBs in the environment. On the basis of this data, EPA stated that it should ban use of PCBs in non-enclosed systems to level off the emissions in the ecosystem. There was no calculation of what the expected levels of PCBs would be with or without the ban. Since PCBs are suspected carcinogens, and EPA policy is that there is no safe level for carcinogens in the environment, there is an incentive for EPA to demonstrate carcinogenicity rather than a widespread range of benefits.1/

On the other hand, when EPA regulated the non-essential aerosol uses of chlorofluorocarbons (CFCs), it used the results of a National Academy of Sciences study which included models for the depletion of the ozone layer over the next 50 years at the current levels of CFC emissions. From this model can be predicted the potential exposure to ultraviolet rays (which have been linked with skin cancer). A very complete model for the dose response relationship of human sensitivity to ultraviolet rays is available; EPA estimated the numbers of skin cancers expected per year due to current levels of CFC use and the number expected per year with a total ban on CFCs. Thus, the benefit of a total ban on CFC use was expressed in numbers of cases of skin cancers prevented per year.

The Clean Air Act under section 112 regulates hazardous air pollutants which cause illness or premature death; the five chemicals for which standards have been published or proposed all have been demonstrated to have adverse health effects. The quality of data available to estimate these effects varies for the different chemicals and is discussed in section 2.3.2; the vinyl chloride standard and the proposed benzene standard show an actual calculation of risk reduction. For instance, when EPA set the emissions standards for vinyl chloride under Section 112 of the Clean Air Act, it calculated that the number of cases of cancer in the population in the area of plants manufacturing vinyl chloride would be reduced by a range of one to 19 cases per year.

To summarize, in only a few cases of regulations now in effect (including trihalomethanes (SDWA), CFCs (TSCA), vinyl chloride (CAA)) has EPA specifically quantified the actual improvement in health to be brought about

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1/The PCB rule may not be a representative TSCA regulation since Congress wrote the ban of PCBs into the Toxic Substances Control Act; with this mandate, it may have been a misallocation of resources to collect data on benefits of control rather than to concentrate on rapid implementation of the ban.

by the regulation. If these estimates are accurate and reflect current knowledge, they can be used directly in the cost-effectiveness analysis. (See the case studies of vinyl chloride and CFCs in Appendices B and C.)

However, since EPA has not routinely conducted risk reduction analysis for all regulations, we will not be able to take the calculation directly from support documents; in some, we will have to supplement the data presented. Even in the three regulations discussed above, the risk reduction was calculated for only one of the health benefits. For many of the substances, there is a range of toxic effects and thus a range of benefits to be derived from the regulation. In addition, there are examples of other risks which are not routinely evaluated for EPA regulations:

- Delayed Health Effects. These include mutagenicity which may affect future generations and are usually excluded from risk reduction analyses.
- Decreased Food Chain Exposures. These are not always included as potential benefits of a regulation. EPA is required to consider the food chain when establishing toxic pollutant effluent standards. However, most of the other regulations are not expressly required to consider this risk, and hence, it is not evaluated.
- Synergistic and antagonistic interactions. Scientific studies show that some chemicals work together to cause human health effects, and sometimes one chemical acts to increase or decrease the human health risk of another chemical. The chemical-by-chemical approach to risk reduction assessment hinders consideration of these synergistic and antagonistic human health effects; however, they should be considered to thoroughly evaluate the benefits of a regulation. Unfortunately, information about chemical interactions is limited.

2.3.2 Quality of the Benefits Data

We have surveyed EPA regulations of toxic pollutants to determine how the benefits of a proposed regulations are characterized and have found there to be various inconsistencies in the types of benefits information available. In this section, we discuss the way EPA has approached the measurement of benefits with examples where this has been partially or fully done. We then point out the data required to complete the measurements and the problems involved in obtaining it.

A. Inconsistencies in the Measurement of Effects

Benefits of pollution abatement have been expressed in a variety of ways: reduced number and severity of health effects, reduced ambient concentrations, and decreased discharge levels. For comparisons to be useful, risk reductions from various regulations should be expressed in comparable units of measurement. Where they are not, it is necessary to apply a uniform methodology to convert them. For example, to convert benefits measured in decreased exposures to benefits measured in decreased health effects may involve gathering information on the relationship between decreased exposures and resulting health effects (e.g., incidence rates for leukemia at various exposure levels).

Even where risk reductions are already expressed in terms of health effects, further evaluation of the effects is necessary. Risk reductions have been measured in terms of number of lives saved, number of cancers prevented, or other specific health changes. One type of cancer or health problem might result in premature death, where another type might be curable. Society might place quite different values on preventing one over preventing another.

B. Calculation of the Benefits of a Regulation

Where EPA has presented a very comprehensive risk assessment analysis, we can analyze the data relatively easily. For instance, when we know the risk at the baseline and resulting exposures, and the total number of individuals in the exposed population, we can calculate the reduction:

$$\text{Risk reduction} = (R_0 - R_1) \times (\text{Number of individuals exposed})$$

where R_0 is the baseline risk
and R_1 is the reduced risk resulting from the regulation.

In most cases, however, some of the data needed are not available or are highly uncertain. Data are often missing in both areas: the amount of risk reduced and in the number of people exposed.

a. Amount of Risk Reduced

In order to estimate the health benefits of the regulations, we have to know how humans respond at the different levels of exposure. This information can be derived from a dose-response relationship, but we often do not have this information for human exposure. Information of this type is sometimes available from workplace studies, where the environment can be monitored more easily than for the general population, or from clinical investigations. If so, we must decide if the data are directly transferable to an environmental regulation, or if they must be adjusted to be more comparable to the target population (e.g., the healthy male worker vs. the general population). More often, the exposure data comes from laboratory experiments on animals, where exposures are easier to control and monitor. In these cases, we must convert the animal exposure data to human exposure data. In doing so, we make several assumptions:

- That the type of response is the same in animals and in humans. (Animal carcinogenicity is an indication of human carcinogenicity, for example.)
- That the differences in route of exposure, where not the same, can be taken into account.

- That we can convert the data based on some factor (for example, exposure per weight or exposure per body surface).

- That where experiments were done with high dosages, we can adopt a model to extrapolate to the relevant dose. The levels found in the environment are almost always much lower than those used in the laboratory.

EPA has developed the methodology for treating each of these assumptions in the case of carcinogens. Each of these assumptions may introduce error into the estimates. Assumptions should be clear and the range of estimates should be carried throughout the analysis. We have found that the data necessary to make these calculations is generally available for EPA toxics regulations. Most of the chemicals have undergone extensive laboratory research, and the data has been collected in criteria documents developed by EPA and the National Institute for Occupational Safety and Health (NIOSH).

After the dose response relationship is worked out, the risk at the various levels of environmental exposure must be predicted. To do this, it is necessary to know what the levels of exposure are both before and after the regulation goes into effect. This can become quite complex, because it is necessary to predict what will happen in the future both with and without the regulation. This involves measurement of current levels and anticipating what future production, distribution and behaviors will be. Two different problems which make this difficult are:

- Baselines. What would industry (and government) have done in the absence of regulation?

- Stocks and Flows. Often the response to regulation requires a finite amount of time, during which stocks and flows of benefits change slowly in reaction not only to the regulation but also to changes in technology, market effects, and the physical or biological properties of the system under study. Keeping track of each stock and flow over time is difficult, and selecting an appropriate time frame for analysis becomes important. (For example, do benefits continue to accrue forever; if not, how long?).

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[2] Yocum, J. et al. "Indoor/Outdoor Air Quality Relationships" Journal of the Air Pollution Control Association 21: 251-259, May 1971; Wade, W. A. et al. A Study of Indoor Air Quality, U.S. EPA Report No. EPA 650/4-74-042, September 1974; Hershaft, A., Morton, J. and Shear, G. A Critical Review of Air Pollution Dose-Effect Facts, NTIS Publication No. PB 251 519, 1976.

[3] The Carcinogen Assessment Group's Method for Determining the Unit Risk Estimate for Air Pollutants. EPA, July 31, 1980.

In some cases even the current exposure levels are not known. For instance, in the drinking water standards that set maximum containment levels for sixteen contaminants, the objective was to insure that no drinking water supply contained these chemicals above the maximum contaminant level (MCL). It was known that these chemicals were found as contaminants in drinking water supplies in various areas of the country, but there was not a systematic survey of their distribution throughout the country. It might be assumed that a system in compliance had lowered the concentration to the level of the MCL, but this would be inaccurate since individual contaminants were not found in some supplies--that is, they were below the MCL before the regulation went into effect. Without a baseline with which to compare, and with very inaccurate assumptions on which to base the exposure levels achieved, one would have difficulty in assessing changes in levels of exposure. However, there might be more data for individual contaminants. (See the discussion of the Safe Drinking Water Act standard for lead in Appendix A.)

The TSCA PCB regulation presents a similar problem. It is well known that PCBs are ubiquitous and that they are essentially not biodegradable; they become a permanent part of the ecosystem. However, there is no measurement of the current levels of exposure to PCBs in the United States. An estimate of average exposure might be made, but there are cases where PCB contamination has resulted from spills or leaks. Risks from these sources would not be captured in an overall average estimate of background PCB level.

In these two cases, much work would have to be done to gather the data or develop models to estimate the differences in exposure as a result of the regulations. However, in most cases, EPA has already made these estimates.

b. Numbers of Individuals Affected by the Regulation

In order to express the risk reduction in terms of lives saved, cases of disease prevented or total quality of lives improved, we need to know how large a population the regulation affects. If we are dealing with a widely distributed pollutant, we might assume that the whole U.S. population is at risk. However, a chemical which shows a fetotoxic effect or is primarily hazardous for the elderly will affect only a specific proportion of the population. Others such as ambient lead, are regional in effect.

c. Summary: Data Available to Estimate Benefits

As pointed out above, in order to measure the effects of a regulation, we must have information on the risk at various levels, the change in exposure brought about by the regulation and the size of the population affected by the regulation. In Exhibit 2-2, we have summarized availability of data for the various EPA toxics regulations we surveyed. We looked at regulations under the Clean Air Act, Section 112; Safe Drinking Water Act; Federal Water Pollution Control Act, Section 307; TSCA, and cancellations in effect under FIFRA, to see if the data are available in a form that is readily usable by our methodology, whether revision and extrapolation and assumptions about models is necessary, or whether data collection is required.

EXHIBIT 2-2

AVAILABILITY OF DATA NEEDED FOR PERFORMING RISK REDUCTION ANALYSIS
(This applies to the major health benefit affected by a regulation)

Regulation		Risk Reduction Analysis	Risk Data Available		Level of Toxic Exposure		Population Exposed
			Human Dose Response	Animal Dose Response	Baseline	Result of Regulation	
Clean	Asbestos	0	+	0	+	+	-
Air Act	Berylluim	0	+	0	+	+	-
Section	Mercury	0	+	+	+	+	-
112:	Vinyl Chloride	++	+	++	++	++	++
	Benzene	++	+	-	++	++	++
Clean	SOx	0	+	0	+	+	+
Air Act	Particulates	0	+	0	+	-	+
Section	NOx	0	+	0	+	+	-
109:	CO	0	++	0	+	+	+
	Lead	++	++	0	++	+	++
	Hydrocarbons	0	++	0	+	+	-
	Ozone	0	+	0	+	+	+
FIFRA:	Aldrin/Dieldrin	0	+	+	0	0	0
	DDT	0	+	+	0	0	0
	DDD	0	+	+	0	0	0
	Mirex	0	0	0	0	0	0
	Endrin	+	0	+	+	+	+

++ = Data in usable form.

+ = Data need updating, revision or extrapolated.

- = Data collection necessary.

0 = Existence of these data in any form has not been confirmed.

EXHIBIT 2-2

AVAILABILITY OF DATA NEEDED FOR PERFORMING RISK REDUCTION ANALYSIS
(continued)

Regulation		Risk Reduction Analysis	Risk Data Available: Human Dose Response	Risk Data Available: Animal Dose Response	Level of Environmental Exposure: Baseline	Level of Environmental Exposure: Result of Regulation	Population Exposed
SDWA:	Arsenic	0	+	++	-	-	0
	Barium	0	+	0	-	-	0
	Cadmium	0	+	+	-	-	0
	Chromium	0	+	++	-	-	0
	Fluoride	0	++	0	-	-	0
	Nitrate (as N)	0	+	+	-	-	0
	Selenuim	0	+	++	-	+	+
	Silver	0		0	-	0	0
	Mercury	0	-	-	--	-	0
	Lead	0	++	++	+	+	+
	Endrin	0	0	0	0	0	0
	Toxaphene	0	0	0	0	0	0
	Lindane	0	0	0	0	0	0
	Methoxychlor	0	0	0	0	0	0
	2,4,5 T	0	0	0	0	0	0
	Trihalomethanes	++	+	++	++	++	++
	Radionuclides	++	++		++	++	++
TSCA:	CFCs	+	++	0	+	+	+
	PCBs						
FWPCA 307:	Aldrin/Dieldrin	0	+	+	+	+	0
	Benzidene	0	+	+	+	+	0
	DDT	0	+	+	+	+	0
	Endrin	0	0	+	+	+	0
	PCB	0	0	+	+	+	+
	Toxaphene	0	+	+	+	+	0

++ = Data in usable form.

+ = Data need updating, revision or extrapolation.

- = Data collection necessary.

0 = Existence of these data in any form has not been confirmed.

We found that under the Clean Air Act, the vinyl chloride and benzene regulations can be evaluated with the data available. The other regulations would require some extrapolation, with data collection necessary on the population exposed for five regulations. The FIFRA regulations require more work in collecting data. Since these pesticides were cancelled before the new procedures for the RPAR process was initiated, the economic analysis is not as thorough as is now required. Thus, these would probably not be good candidates for this analysis. Although there have been pesticides whose cancellations have been proposed under the new procedure, hearings are in progress or still to be held on these and this type of analysis would be more valuable if done after final decisions had been made on each one.

There are risk assessment data available on the Safe Drinking Water Act regulations, but determining the levels of exposure will be difficult. This is the same problem that exists for PCBs; significant resources would be needed to gather the data for this part of the analysis.

2.3.3 Treatment of Joint Benefits

When evaluating the benefits of a regulation, it is necessary to know specifically what benefits can be attributed to that regulation. In some cases, a regulation has wider benefits than originally intended. Joint benefits are applicable to technology-based standards. For example, the safe drinking water regulations require installation of specific controls to prevent a particular pollutant from entering drinking water. One control usually operates to prevent several pollutants from being discharged and thus, one control leads to several benefits. To accurately reflect the benefits of a regulation, all benefits accrued must be evaluated.

Another example of a joint benefit is the mobile source lead emission standard. The use of fuel without lead additives substantially decreases lead and particulate emissions. In an analysis of the benefits of the lead standard, risks reduced from controlling particulate emissions, in addition to lead emissions, must be evaluated. Joint benefits must be evaluated on a case-by-case basis.

2.3.4 Technological Innovation Due to New Regulations

Regulations can stimulate technological innovation. This is particularly so when existing control technology is not capable of providing the necessary level of control required by a new regulation. This innovative technology can consist of process modifications or improvements in add-on control equipment.

Innovative process modifications can entail materials substitutions or changes in the operating procedures of a process. The result is often a reduction in the toxicity of the chemical by-products as well as the quantity of the discharges.

Improvements in add-on control equipment can entail either improving the control efficiency of existing equipment or developing new equipment and procedures. Control equipment innovations also include developing new applications of existing control methods.

Both process modifications and improvements in control equipment have the direct benefit of reducing the amount of discharge of a substance. They also have indirect benefits. Process modifications can result in increased process productivity, increased labor productivity and decreased operation and maintenance costs. Once a process modification or improvement in control equipment is developed, it may have application to other processes as well.

Although the potential for technological innovation due to a regulatory action exists, it is very difficult to fully assess the potential benefits. This problem arises from the difficulty in trying to identify possible future innovations and then predict and quantify their relative benefits. However, this potential benefit of a regulation must be identified and analyzed to the best degree practicable as part of the benefits assessment.

2.3.5 Valuation of the Benefits of Regulations[4]

The conversion of physical measures of effect to measures of social value of those effects is a controversial issue in evaluating the benefits of a regulation. The motivation for performing this conversion is to develop comparable measures of impact so that all the effects of different regulatory strategies can be compared.

One approach has been to convert all effects into terms of their monetary value. This involves the question of evaluating the dollar value of a life saved or prolonged, the costs of treating a disease caused by the pollutant or the foregone economic activities of the deceased. Further difficulties develop when considering the question of quality of life.

There is no general concensus that human health impacts can be monetized satisfactorily by any currently available economic methodology (such as willingness to pay); however, to produce adequate measures of social benefits, it is possible to develop other measures that can be useful in comparing health effects. These include lost-life years, extent of disability, as measured in terms of lost employment or lost activity, and days hospitalization. In our preliminary analyses, we have not attempted to evaluate health in these terms. Our policy, in the absence of satisfactory monetary measures of benefits, has been to go as far as possible with physical units of damage, and be conservative about applying any kind of economic valuation model. To do otherwise is to produce estimates of benefits that depend heavily on implicit assumptions about social valuation. We have expressed the benefits in terms of cancers prevented or blood lead levels reduced.

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[4] For additional discussion of these issues, see Benefits of Environmental, Health and Safety Regulations, report to the Committee on Governmental Affairs, United States Senate by the Center for Policy Alternatives at the Massachusetts Institute of Technology, U.S. Government Printing Office, March 25, 1980.

2.3.6 Comparing Costs and Benefits in Different Time Periods

The issue of whether or not and how to discount streams of costs and benefits accruing over time as a result of public investment decisions remains a controversial aspect of cost-effectiveness analyses.[5] The application of discounting to monetary values (e.g., streams of dollars) is fraught with difficulties. Discounting nonmonetary quantities (e.g., numbers of cancers) is even more problematic. Assumptions regarding the relative value of benefits and costs in different time periods are very important because the results of cost-effectiveness analyses of the control of toxic substances are often very sensitive to discounting procedures, since the benefits of control generally accrue in the medium to far-off future. We do not intend to resolve the methodological or philosophical discounting issues. Instead, the importance of the topic requires that all the assumptions regarding the use of discounting, and the implications of the assumptions, be spelled out clearly.

The use of discounting to compare values in different time periods is often obscured in discussions of difficulties in measuring values in different time periods. "Measuring" problems should be kept separate from discounting problems. Two measuring difficulties, uncertainty and changes in valuation with wealth, will be touched on briefly, and then our approach to discounting will be discussed.

Uncertainty of future outcomes is often used as a justification for valuing the future less than the present. Proponents of this argument have stated that since we are uncertain of future events, they should not bear heavily on current decisions. Uncertainty, however, should be addressed separately and, if need be, a range of possible outcomes should be described with appropriate probability assessments.[6] Therefore, the choice of whether to discount should be separated from problems of uncertainty.

The fact that individual valuations of unique or irreplaceable assets (e.g., a life) have been demonstrated to vary with changes in wealth,[7] has also been construed as affecting the choice of whether to discount future outcomes. Changes in the valuation of reductions in risk by future, presumably more wealthy, generations should be assessed explicitly, outside the choice of a discount rate. If streams of benefits or costs are left in their natural units (e.g., cancers), it is difficult to "revalue" them according to the preferences of future generations. The discount rate can be adjusted to account for this; however, this should be done separately from a choice of appropriate discount rate. If benefits or costs are in dollars, they can be adjusted up or down to reflect assumptions about the preferences of future generations.

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[5] Decision Making for Regulating Chemicals in the Environment, National Academy of Sciences, Washington, D.C., 1975, p. 177, and "Procedures for Valuing Lives" by Richard Zeckhauser, Public Policy, Vol. 23, No. 4, p. 438.

[6] The use of subjective inference is not universally recognized. See Chapter 13 of Games and Decisions by Luce and Raiffa (Wiley, 1957) for a discussion of "objectivist" versus "subjectivist" schools of statistics.

[7] "Demand for Insurance and Protection," by Philip Cook and Daniel Graham, Quarterly Journal of Economics.

The choices of whether or not to discount and how to discount should reflect "society's rate of time preference." Prevailing rates in existing capital markets are generally used as a guide to discounting monetary values. Even the application of these rates to public investment decisions can be seriously questioned. Discount rate choice involves such philosophical issues as whether "society's choices," as reflected through government decisions, should reflect personal preferences only, or whether the state is more than just a collection of individuals and has an existence and interests of its own.[8] Also, the absence of markets for contingent claims and the unability of government to transfer the advantages of risk spreading to private investors results in erroneous signals of time preference from existing imperfect capital markets.[9]

It cannot be shown unambiguously whether signals of time preference from imperfect capital markets should be adjusted up or down when applied to public investment decisions. This is not to say that existing rates in capital markets should be blindly applied as the correct rates. Instead, the impossibility of knowing the "true" rates is generally recognized and sensitivity analysis is utilized to reveal the complications of using different discount rates (within a reasonable range).

The difficulties in discounting irreplaceable assets (e.g., lives) are even greater because there are no market signals at all from which to infer the appropriate discount rate to use.[10] The choice of how to value streams of benefits and costs of this type must ultimately rest with the policymaker. Therefore, at a minimum, the streams of benefits and costs that are expected to accrue over time as the result of policy decisions should be spelled out fully and shown in their appropriate time periods. Additionally, however, the implications of different assumptions regarding the time preference for these types of benefits and costs should also be shown.[11] When, for example, numbers of cancers prevented are discounted to present value at a discount rate, the actual number of cancers prevented does not change, but the opportunity costs for the investment required to prevent the cancers is taken into account. If, in any time period, monetary resources can be used to save lives and prevent cancers, then this opportunity cost of investments made today should be reflected when valuing lives saved and cancers prevented in the future.

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[8] "Uncertainty and the Evaluation of Public Investment Decisions," by Kenneth Arrow and Robert Lind, American Economic Review.

[9] Uncertainty and the Evaluation, op. cit.

[10] Demand for Insurance, op. cit.

[11] Decision Making for Regulating, op. cit.

Although this argues for the necessity to discount, it does not provide a basis on which to choose a discount rate. Until one is derived, it is appropriate to use sensitivity analysis and to compute benefits and costs over a range of social discount rates, including a zero rate, and to see if the outcomes over the range substantially influence the magnitude of benefits in relation to costs.[12/] This procedure highlights the implications of different discounting assumptions and is consistent with the approach used to discount a stream of monetary benefits and costs. Therefore, sensitivity analysis of discount rates is illustrated in our case study of chlorofluorocarbons (Appendix C).

2.4 SUMMARY

The benefits of toxic substance regulations can be assessed and combined in different ways. It may not always be possible or even necessary to specify benefits in terms of human health effects. Other benefit measures may be used. Our review of selected EPA regulations has emphasized several key issues:

- Risk assessments are often incomplete and do not specify all relevant environmental or health impacts.

Therefore, an initial task in studying the benefits of an EPA toxics regulation will often be to broaden existing risk assessments using available data and accepted models or assumptions.

- Risk reduction assessments have rarely been performed and are not always specified for all risks addressed by a regulation.

Thus, in most cases, one must perform a risk-reduction assessment whether or not a baseline risk assessment is already available. In many cases, both types of assessment will need to be done.

- Regulatory standards and benefits are often defined in terms of different categories of environmental impacts, thus hindering comparative analysis.

Different regulations address the use of chemicals, their discharge into the environment, levels of pollution present in air and drinking water, human exposure, and health and welfare effects. Thus, the benefits are often defined in very different terms. In order to do a comparative analysis, one

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[12/] *Ibid.*, p. 178.

must standardize the benefits to common types of impacts, not necessarily (but preferably) health impacts.

Other practical and theoretical problems must be handled on a case-by-case basis. For example, the problem of joint benefits resulting from a control requirement may not be relevant for all regulations. The theoretical problem of aggregating disparate benefits of a single regulation into one measure for cost-effectiveness analysis may also be amenable to practical resolution in many cases.

3. COST ISSUES

3.1 INTRODUCTION

The ability to make meaningful cost-effectiveness comparisons across EPA regulations depends to a large extent on the availability and comparability of estimates of costs of EPA regulations. This chapter addresses issues important to comparing estimates of costs of EPA regulations.

Pertinent cost issues were initially identified through a brief review of major EPA toxics regulations. That review yielded a list of cost issues which seemed to require further analysis. Refinements to the list were made as the case studies, discussed later in this report, were conducted. The final list of issues addressed is:

1. Quality of cost estimates;
2. Joint costs, common costs, and economies of scale; and
3. Nonmonetary compliance costs.

The remainder of this chapter is focused on these three cost issues. First, each issue is first fully described. Next, the extent to which the issue is relevant across regulations and the seriousness of the problem it presents are discussed. For those issues found to be potential barriers to successful cost-effectiveness analysis of EPA regulations, alternative solutions are identified and discussed.

3.2 DISCUSSION OF ISSUES

3.2.1 Quality of Cost Estimates

Description

The total cost of a regulation can be described as its social welfare cost to society. "Social welfare cost" means the net impact a regulation has on the welfare of society exclusive of the risk reduction effects discussed in the previous chapter. The welfare losses to society as a whole can be classified into three types:

- Changes in consumers' and producers' surplus caused by the direct costs of compliance;
- Transactions costs imposed by unanticipated changes in employment patterns, investment schedules, relative prices, and the competitive structure of industries; and
- Externalities not captured in existing markets or regulations.

In addition to these costs, resources may be "redistributed" or transferred between industries, regions, or subgroups of the population. The imposition of costs on one particular industry will shift demand and income away from the industry on which costs are imposed toward other industries. People employed in the regulated industry will lose income to people employed in other industries. Because no industry is evenly distributed across all regions and communities of the country, inter-industry shifts will result in inter-community and inter-regional shifts. Insofar as these redistributions occur, they may impose transactions costs. The amount of the resource transfer itself is not counted as a cost.

In any given cost analysis done in support of a regulation, all or some of these costs may be analyzed. With this framework, we can discuss the quality of available cost estimates for EPA toxics regulations. The issue of quality has three dimensions:

- are all the costs included which should have been included;
- are they properly identified (e.g., changes in consumers' surplus or transactions costs); and
- is each category of cost estimated properly.

Determining whether all costs have been included that should have been included is relatively straightforward. The task of developing a detailed list of costs is not complicated nor is checking that the costs have been properly identified.

The most important of the three dimensions, and the most difficult to evaluate is the third: whether each category is properly estimated. A methodology to estimate the costs of regulation has two major segments: the specification of the appropriate baseline and the comparison of costs introduced by the regulation to the baseline. In general, there are three choices for the baseline:

- current standards;
- current practices; and
- expected behavior in the absence of the standard.

The third choice, expected behavior in the absence of the standard, is the theoretically correct baseline. However, using this baseline opens the cost analysis to additional uncertainties associated with predicting expected behavior. One issue involves current standards as a baseline rather than current practices. In this case, the costs of meeting current standards (where current practices do not meet current standards) are not included in the costs of the regulation. This avoids double counting, since presumably the costs of attaining current standards were counted when the current standards were formulated.

In the economic analyses we reviewed, current practices were used as the baseline in all cases. However, in all cases, there were no existing regulations. Therefore, there are no current standards distinct from current practices. The important aspect of this issue is that the baseline assumptions must be consistent across all the regulations being compared.

Another aspect of whether costs are estimated properly is the effect of errors and uncertainty. Errors and uncertainty can enter the analysis at each stage of the estimation procedure. By the end, their accumulation can be so great as to make the estimates virtually worthless. The basis of all of the cost estimates for each regulation is necessarily some type of engineering study of the capital and operating expenses of controlling pollution.[1] If the control process is already being used, or if it is an extension of well-known technology, the cost estimate may be quite accurate. However, even if the process is well known, unexpected delays or other problems in installing the equipment can make cost projections obsolete. If the control technology is highly innovative, capital and operating cost estimates are even more uncertain. It is important to note that one evaluation of the accuracy of industry and EPA forecasts of pollution control costs showed that industry's predictions were off by an average of 52 percent, and EPA's predictions were off by an average of 43 percent.[2]

Indirect costs are the result of the imposition of capital and operating costs, and any error or uncertainty in estimating these costs will also influence the estimates of these other costs. In addition, further errors in the estimation of other costs will be generated by errors in assessing the linkages between capital and operating costs, and other costs. For example, an estimate of consumer surplus lost as a result of pollution control expenditures is a function of both the amount of capital and operating costs and the elasticity of demand. Errors in estimating either of them will contribute to errors in the estimate of lost consumer surplus. Many of the distributional effects are in turn functions of lower demand for a product. The errors in estimating the linkages between demand and these other costs (such as lost employment and income) create further errors in estimating the other costs.

In many cases, it may be difficult to judge the quality of the estimates because the uncertainties associated with the initial cost estimates and the linkages between costs are not made clear. In these cases, the assessment of quality is itself uncertain. Measures of uncertainty (e.g., variance or providing a range of estimates rather than point estimates) are useful if the quality of the estimates are to be properly assessed.

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[1] Capital and operating costs of controlling both the pollutant being regulated and pollutants generated through "inter-media effects" should be included. Inter-media effects are discussed further below.

[2] Putnam, Hayes, and Bartlett, Inc., "Comparisons of Estimated and Actual Pollution Control Capital Expenditures for Selected Industries," unpublished, June 1980, p. 6. The numbers cited here are the average of the difference between estimated and actual costs. If a range of estimated costs are given in the report, the midpoint of the range is used.

Review of Existing Economic Analyses

Cost estimates are assumed to be of good quality if they reflect:

- Knowledge of the linkages between the capital and operating costs (direct costs) on the one hand, and the remaining transactions and social welfare costs (indirect costs) on the other; and
- Good estimates of the capital and operating costs, from which all other costs derive.

The following discussion of cost estimates for EPA toxics regulations is organized around many of the EPA statutes pertaining to toxics.

Federal Water Pollution Control Act, Section 307. The analyses done in support of regulations issued under Section 307 of the FWPCA present estimates of the capital and operating costs of complying with the regulations. Quality of the estimates is difficult to assess because an explanation of how those estimates were derived is not given in the documents we reviewed. The analyses acknowledge that there may be impacts on industry concentration and on those who use the product as an input to further processing, and provide quantitative estimates of the latter impact. However, the linkages between the original industry and the users of its products are not specified, and the derivation of the cost to users is not described. The distinction between social welfare costs and distributional effects is often made in the analyses, but transactions costs are not estimated.

Toxic Substances Control Act. In estimating the cost of restrictions on the use of PCBs, EPA considered both the transitional and long-term costs of restrictions on the use of PCBs.[3] Transitional costs include the costs of decontaminating or scrapping existing equipment containing PCBs, and the ban on rebuilding PCB transformers. These costs cease being incurred when the use of existing PCBs is ended. Long-term costs include the additional cost of non-PCB capacitors and transformers. These costs continue into the indefinite future. Transitional and long-term costs, the direct capital and operating costs of the PCB restrictions, are estimated in some detail by EPA, with substantial input from industry; fifteen different uses of PCBs are individually examined in deriving these estimates. In general, the assumptions and uncertainties are presented in considerable detail, and the estimates appear to be as precise as could be reasonably expected. Estimates of the expected cost, as well as likely ranges around that estimate, are given; therefore, the amount of uncertainty is quantified. While occasionally other types of costs are alluded to, no estimates are made of these costs. The distinction between social welfare costs and distributional effects is not made in the analysis, but all of the costs estimated are social welfare costs.

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[3] EPA, PCB Manufacturing, Processing, Distribution in Commerce, and Use Ban Regulation: Economic Impact Analysis. (Washington, D.C.: EPA, 1979) pp.27, 73.

EPA has chosen to regulate chlorofluorocarbon (CFC) emissions in two stages: nonaerosol applications and aerosol applications. The analysis of the cost of the proposed nonaerosol regulation estimates compliance costs in each of seven product areas, plus miscellaneous applications.[4/] For each product area, the analysis estimates the costs to industry and consumers of various control options. For each control option considered, the capital and operating costs are estimated. Details of the estimation procedure are not given, so it is difficult to determine the quality of these estimates. Impacts on total employment and consumers are also estimated, although the estimates are stated in an imprecise fashion (i.e. "final product prices would probably rise by no more than one percent for furniture products and by much less in the transportation markets"). This imprecision seems to be an effort to take into account the uncertainty associated with the estimates of direct compliance costs and the linkages between those costs and unemployment and price effects. Where quantitative cost estimates are provided, only point estimates are given.

The analysis makes a clear distinction between social welfare impacts and distribution effects.[5/] In addition to estimating the social welfare costs, the analysis also attempts to assess how these costs are distributed among different segments of the industry, and divided between producers and consumers. Again, these estimates are relatively imprecise, reflecting the uncertainty associated with their derivation.

In the analysis of the cost of the existing ban on aerosol uses of CFCs, the economic impact of the regulation is taken to be the aggregate change in value-added by industry, with value-added being defined as the difference between sales revenue and outside purchases of raw materials or components from other industry sectors.[6/] This definition is not a measure of change in economic welfare for society as a whole, though it is an indicator of the impacts on the industry immediately affected by the regulations. The fact that value-added losses in one industry may be offset by increases in value-added in other industries is acknowledged in the analysis. However, those offsetting gains are specifically excluded from consideration.[7/] Demand equations are estimated and consumer surplus is estimated, but the

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[4/] EPA, Economic Implications of Regulating Chlorofluorocarbon Emissions from Nonaerosol Applications (Washington, D.C.: EPA, 1980), p.62.

[5/] Ibid., pp.16-18.

[6/] International Research and Technology Corporation and Policy Models, Inc., The Economic Impact of Potential Regulation of Chlorofluorocarbon-propelled Aerosols, (Washington, D.C., EPA, 1977), pp. 4-1,2.

[7/] Ibid., p.4-3.

estimates do not seem to be very precise.[8] A process model is used to estimate the effects of restrictions on the use of chlorofluorocarbons on the markets for precursor chemicals and byproducts of the production process. The model is not fully described in this analysis, so it is impossible to assess its quality. In general, the discussion of the various economic impacts is not integrated into a coherent whole.

Safe Drinking Water Act. The cost of controlling trihalomethanes (THM) in drinking water is estimated somewhat differently than the other cases discussed here. Water systems which exceed the THM standards have three options available to satisfy the standards. In order of increasing cost, they are:

1) modifying chlorination or other treatment procedures;

2) changing disinfectants; and

3) adding an adsorbent.

In order to estimate total national costs, the number of water systems likely to select each of the treatment options must be established. This is determined by building a decision tree, in which a logical sequence of decision points is designed to distribute the water systems along the path of pollution control decisions and expenditures each would most likely follow. Once the distribution of treatments is established, an engineering process model is used to estimate the capital and operating expense of the national program. Only these direct compliance costs are estimated; distributional costs are not considered. However, the analysis does discuss the effect of the regulations on the products used in controlling THM; in general, the effects are thought to be minimal. Because the model is not presented in full detail, it is difficult to determine if the assumptions are valid, but the approach seems to be thorough. Only point estimates are given; no measures of uncertainty are discussed.

Clean Air Act. Benzene emissions standards have been proposed under Section 112 of the Clean Air Act. So far, EPA has completed cost analyses for two proposed standards: emissions from storage tanks and emissions from maleic anhydride plants. For storage tank emissions, the capital and operating costs of several different types of controls are estimated from conversations with vendors of such equipment. These costs are then used in a financial model of benzene producers and consumers to determine the decreases in return on investment. Impacts are estimated separately for large and small plants. The analysis also includes a discussion of the impacts on foreign trade, industry output, employment, and growth, and on the suppliers of emissions control equipment. Because the equipment used to control emissions is already available, the estimates of capital and operating costs are probably relatively accurate. The financial model is developed from FTC data for the chemical industry and petroleum and coal products industry as a whole. Therefore, it is not clear how closely this approximates that subset of the industry which produces and consumes benzene. The discussion of the other impacts is quite cursory, although the brevity of the discussion is justified by the fact that the impacts are quite small. In most cases a point estimate rather than a range is given.

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[8] Table B-9 on page B-14 contains estimates of "willingness to pay" (the area under the demand curve), estimated expenditures, and actual expenditures for 1971. Although the derivation of estimated expenditures is not explicitly discussed in the text, it appears to be the product of actual quantities times the price estimated from the demand equations, assuming the estimated elasticities to be true. The estimated expenditures are usually substantially different from the actual expenditures, casting doubt on the elasticity estimates, which in turn cast doubt on the estimates of willingness to pay.

In the analysis supporting the proposed regulations on emissions from maleic anhydride plants, estimates of capital and operating costs were associated with two different control options. Both point estimates and ranges are given. Effects on employment, plant closures, and energy usage are derived from these estimates. Although the effects on small business are not explicitly taken into account, one of the plants threatened with closure by the regulations was a small business.

Vinyl chloride has also been regulated under Section 112 of the Clean Air Act. The economic analysis provides estimates of the capital and operating costs for several different control options. From the estimates of capital and operating costs, price and rate of return impacts are derived. These results are used to examine each plant in the industry, and determine if that plant is likely to close due to the imposition of controls. Employment and balance of trade effects are also discussed.

The equipment costs are estimated by obtaining quotations from vendors. Because all of the control technologies incorporate existing technologies, the quotations are probably relatively accurate. The predictions of the investment and pricing behavior of firms are quite uncertain, as is acknowledged in the study.

Under Section 111 of the Clean Air Act, EPA is to issue regulations establishing performance standards for new sources of air pollution. These New Source Performance Standards (NSPS) are set on an industry by industry basis. Formal economic assessments have been performed by EPA for each industry regulated. An example of these is the analysis of the proposed regulations controlling the fugitive emission of volatile organic compounds (VOC) by the synthetic organic chemicals manufacturing industry. (Fugitive emissions are unplanned leakages from plant equipment.) The analysis begins by determining what type of control equipment is needed for three different model units. Capital and operating cost of that equipment is estimated for three different levels of control. Two scenarios are examined, one in which any cost increases could be fully passed through to consumers, and one in which any cost increases would have to be absorbed by the industry. Under the first scenario, price impacts are estimated, and under the second scenario, the effect on the industry's rate of return is estimated. The price and rate of return effects are then further used to derive the impacts on investments and employment.

EPA estimates that the net costs for two of the three control options analyzed are negative; i.e., the firm actually profits from controlling emissions because it reduces the amount of its product lost to the environment. If profits could be made by installing pollution control equipment, firms might have been expected to have done it voluntarily. However, there are a number of reasons why firms might not have controlled emissions even if it were profitable. The existence of this result does not necessarily cast doubt on the underlying cost data.

Conclusions

Most of the cost analyses we reviewed are of relatively high quality. With one exception, each starts with the proper foundation -- the estimation of capital and operating costs. These estimates are usually obtained from the vendors of the equipment. Although it is impossible to determine the accuracy with which the vendors estimate equipment costs without independent engineering knowledge of the equipment in question, gathering information from the vendors would seem to produce the most accurate estimates possible. Because all of the other indirect impacts are functions of the capital and operating costs, the accuracy of the former cost estimates places an upper bound on the accuracy of the latter.

Where the analysis go beyond the estimation of direct capital and operating costs, the quality declines. Each analysis includes a discussion of a somewhat different list of indirect costs. In addition, the understanding of the linkages between the direct costs and indirect effects such as employment and inflation is generally not shown. In order to forecast these indirect costs accurately, it is necessary to have accurate knowledge of firm investment behavior and consumer demand, and the knowledge of these demonstrated in the analyses is simply not sufficiently documented to indicate that much value should be put in the predictions of indirect effects.

Capital and operating costs are often presented as the amount required on an annual basis over the life of the equipment to support the expenditure. Two different regulations which produce equal annualized expenditures may produce different impacts if the pattern of expenditures over time differs in the two cases. Even if annualized costs are equal, a regulation which imposes large capital requirements at first, and low operating expenditures later, may have a different impact than another regulation with lower capital costs, but higher operating costs. This problem can be resolved by assessing capital and operating costs as two separate numbers, rather than as one combined, annualized rate. Operating costs are usually considered to be constant (in constant dollars) over the life of the equipment, although this is probably a slight distortion of reality. Therefore, all of the information in the stream of operating costs can be represented by a single number. Capital costs are typically incurred over several years, rather than in a single year, so a single number (annualized expenditures or total present discounted expenditures) will not perfectly represent the impact of the expenditure stream. However, since capital expenditures will generally occur over a relatively short period of time and because firms have some flexibility in accelerating or postponing capital expenditures, the true impact will not be distorted too much by summarizing all capital requirements as total present discounted cost.

Although the measurement of capital and operating costs seems relatively straightforward and accurate in the regulations studied, one can easily imagine situations in which such measurement is not so simple. The capital equipment referred to in the regulations discussed above is generally off-the-shelf equipment that could be added on to the existing capital stock. If the requisite technology had to be newly developed, estimates of capital and operating costs would be much less accurate. If the control technology required the renovation or alteration of existing capital equipment and operating procedures, the uncertainty of the cost estimates would also be increased. The more costly and extensive the pollution control requirements, the more likely it is that firms will adjust their entire capital stock or the organization of their operations to perform optimally under the new set of constraints. This change in the production processes makes the true costs of pollution control much more difficult to estimate.

3.2.2 Joint Costs, Common Costs, and Economies of Scale

Description

When conducting any type of cost-effectiveness analysis, it is necessary to determine exactly which costs should be attributed to the regulation under analysis. However, an analysis of one regulation in isolation could prove misleading. The control of one toxic substance may require the installation of equipment that can also be used to control, in whole or in part, other toxic substances. If this is so, it is not immediately clear how the costs of such equipment should be allocated among the several pollutants. Should the entire cost be allocated to the first pollutant which is regulated, or should the cost be divided among all regulated pollutants according to some criterion such as volume?

The issue of equipment which is used to treat more than one pollutant is important because of the way in which chemicals are manufactured and the way in which pollution control equipment operates. According to one analysis of the economic impacts of the Federal Water Pollution Control Act of 1972, "most chemical products are produced in integrated facilities that concurrently manufacture a number of other chemical products, sometimes in excess of 1,000; and the treatment costs for each product are very much a function of the number, type, and production volume of the other products."[9] This means that a distinct set of pollution control equipment will not necessarily be installed to control each pollutant. Instead, pollution control equipment will ordinarily be able to control a large class of pollutants. Thus, the conclusion cited above: the costs of controlling a pollutant depend on how many other pollutants are available to share the cost of the capital equipment and operating expenses.

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[9] International Research and Technology Corporation, Water Pollution Control Act of 1972: Economic Impacts (Washington, D.C.: National Commission on Water Quality, 1976), p. 5-1.

This issue also arises in public utility pricing. Public utilities typically produce several different products (e.g., local calls and long distance calls), and one aspect of the regulation of those utilities is to ensure that the users of each product pay their fair share of the costs, those that are caused by the provision of that product. Typically, some of the costs of producing each product are incurred only to provide that product. For example, the cost of the turbines in a large hydroelectric dam and reservoir are incurred only to provide electricity, not to provide recreation, flood control, or the other outputs of this project. But to the extent that several products are produced by the same piece of equipment, costs are either joint or common. If a piece of equipment may be used to produce two different products, and when producing one product uses capacity that could otherwise be used to produce the other product, the costs are said to be common. When a piece of equipment is used to produce two products in unalterably fixed proportions, the costs are said to be joint. Common costs may be allocated among the products according to a common volume measure or other formula. Joint costs, on the other hand, may not unambiguously be attributed to any particular product; any attribution that is used is essentially an arbitrary one.

It can readily be seen that some of the costs of pollution control will be joint and others will be common. For example, consider a wastewater stream with two pollutants. Assume that the volume of the wastewater stream itself would not be reduced by the removal of one of the pollutants; therefore, the pipes and holding tanks, whose cost is dependent only on wastewater volume, is a joint cost. If both pollutants are removed by filtering, and the consumption of filtering material is dependent solely on the quantity of pollutant rather than the volume of wastewater, the filtering material is a common cost, which can be allocated among the two pollutants.

The conclusions to be drawn from this discussion are: (1) as long as there are joint costs, there is no unique cost for controlling a single pollutant--the cost is dependent on the context within which the pollutant is controlled; and (2) the cost to be measured must be the cost for the program actually contemplated. The first conclusion follows from the definition of joint costs. The second conclusion necessarily follows from the first. The cost of controlling a pollutant has an unambiguous meaning only within the context of a broad program for pollution control. Even within that program, without making an arbitrary allocation of joint costs, we can meaningfully speak only of the costs of the entire program, rather than of the cost of controlling a single pollutant. This is not to say that assessing the cost of controlling a single pollutant is never justified--it simply depends on the context. If pollution control equipment had already been installed, and the program being assessed was one that would control one additional pollutant, the context is the existence of pollution control equipment and the cost is the incremental cost of controlling an additional pollutant (i.e., no joint costs are allocated to the new pollutant).

For purposes of comparing the cost-effectiveness of existing regulations, it is important to know in what context the cost estimates were made, and what assumptions are made about joint costs. For example, cost estimates for controlling a pollutant under statute A may be much greater than controlling it under statute B. But under A, the pollutant may have been the first to be regulated, thereby incurring the cost of facilities that could be used for other pollutants later. On the other hand, under B, the joint costs may already have been incurred when the pollutant in question was controlled, and it may have been allocated only the incremental costs of control.

Review of Existing Economic Analyses

None of the economic analyses surveyed attempt to grapple with the issue of joint costs. A number of the analyses mention the problem, and some note that the costs incurred to control the particular substance under analysis will also control other substances. However, it is assumed that all costs incurred will be charged to the pollutant under analysis.

Conclusions

In one sense, the solution to the problem of joint cost is to realize that there is no solution. If joint costs really do exist, there is no "correct" way of allocating them among the joint products. This implies that one cannot really speak meaningfully of the cost of one product of a joint production process. It only makes sense to speak of the joint costs of the two products taken together. If joint costs are thought to be significant, then joint benefits should also be considered, i.e., the benefits of controlling all of the joint products of the production process. (The significance of joint benefits was discussed in the previous chapter.) If joint costs are not important, it doesn't matter much if they are explicitly included in the quantitative analysis, though they should be included qualitatively. They definitely should not be counted twice. That is, if equipment has already been installed to control one pollutant, and can be also used to control a second pollutant, the costs of controlling the second pollutant should be calculated assuming the equipment is already in place. One way to apply this cost allocation technique is to assign the entire cost of equipment to the pollutant in question but to subtract from this the expected reduction in the marginal costs of controlling other pollutans.

3.2.3 Nonmonetary Compliance Costs

Environmental regulations can impose costs that are difficult to measure and/or value in dollars. A cost is difficult to measure when the effect of the regulation that imposes the cost cannot be predicted ahead of time or measured retrospectively. In some cases, it may be difficult just to assess the magnitude of such effects. A cost may also be difficult to value, even if it can be measured in its own natural units. Leaving costs in their own various natural units does not facilitate cost-effectiveness comparisons between regulations. Some common unit of measure, e.g., dollars, is needed. Two examples of costs that pose valuation problems are balance of trade effects and employment effects. One regulation may have significant balance of trade effects that can be estimated and another regulation may have employment effects that can be estimated. These two regulations cannot be compared for cost-effectiveness, however, until these costs can be compared in some common unit of measure. Transactions costs are often costs of this type.

Despite valuation difficulties, the costs are as real as the direct capital costs of compliance and should be included in the cost-effectiveness analyses of regulations. This section describes how significant we found these costs to be in the regulations we reviewed for this feasibility analysis.

We identified the following eight effects that regulations can have that also impose costs:

- Inter-Media Transfer of Pollutants and Substitution Effects;
- Balance of Trade;
- Employment;
- Plant Closures, Effects on Small Businesses, and Changes in the Competitive Structure of Industries;
- Price Increases;
- Reductions in Productivity and Investment in New Productive Capacity;
- Capital Availability Problems; and
- Effects on Innovation.

We defined each effect and then examined six regulations to assess the magnitude of the costs that these effects impose. The regulations were chosen both on the availability of economic impact data and to include different media and statutes. The regulations examined include:

- Volatile Organic Compounds, proposed New Source Performance Standards under Section 111 of the Clean Air Act;
- Vinyl Chloride, National Emission Standard for Hazardous Air Pollutants under Section 112 of the Clean Air Act;
- Aldrin/Dieldrin, DDT, Endrin, and Toxaphene, Toxic Pollutant Effluent Standards under Section 307 of the Federal Water Pollution Control Act;
- Trihalomethanes, regulated under the Safe Drinking Water Act;

- Chlorofluorocarbons, regulated under Section 6 of the Toxic Substances Control Act; and
- Polychlorinated Biphenyls, regulated under Section 6 of the Toxic Substances Control Act.

Inter-media transfers of pollutants and substitution effects were found to be the most significant of those effects we examined. The vinyl chloride, trihalomethane, PCB, and CFC regulations all had potentially important impacts of this type. In the case of vinyl chloride, contamination was transferred from the air to the land. For CFC, PCB, and THM control, some of the recommended substitutes were found to present hazards that were not adequately addressed in the cost analyses of the regulations. The costs of these hazards should be assessed in a manner similar to that used to assess the benefits of risk reduction.

The other effects were found to be much less important and, in some cases, negligible. Each will be discussed below in detail.

Inter-Media Transfer of Pollutants and Substitution Effects

Some regulations may transfer a pollutant from one medium (e.g., air, water or land) to another medium. Insofar as the pollutant imposes costs when present in the second or receiving medium, these costs must be included as part of the cost of control in the first medium. The pollutant in the second medium may impose costs by affecting human health or the environment. Also, if the pollutant is already adequately controlled in the second medium to protect human health and the environment, the cost of control in the second medium may increase.

The costs imposed by substitutes should also be included in the cost of regulating a substance. For example, if substance B replaces the regulated substance A, the health and environmental effects of substance B are part of the costs of regulating substance A. As above, if the use of substance B is already regulated, the increased compliance cost for that regulation would be part of the cost of controlling substance A. An assessment of the importance of these inter-media and substitution costs is included here with the other nonmonetary costs of the regulations although they could be considered "negative benefits". Since "joint benefits" were included in the previous chapter, we feel that these costs are appropriately considered here.

From the regulations we examined, it appears that inter-media and substitution effects occur frequently. For example, the 1975 EPA support document for the vinyl chloride (VC) regulation (Clean Air Act, Section 112) recommends several control technologies to reduce emissions. One of these technologies, carbon adsorption, would generate VC-contaminated solid waste. The cost of disposing of this waste was not explicitly included in the cost analysis of the VC regulation. More importantly, however, the effects that this waste could have on human health or the environment were not discussed.

Similarly, the control of trihalomethane (THM) in drinking water (Safe Drinking Water Act) may require that chlorine be replaced as the disinfectant in some water systems. The substitutes available, however, also have uncertain adverse health effects (chloramines, chlorine dioxide, and ozone). The potential effect of these substitutes on human health should be considered.

In addition, the full effects of the substitutes for both polychlorinated biphenyls (PCBs) and chlorofluorocarbons (CFCs) controlled under TSCA were not evaluated fully. Some substitutes for either CFCs or PCBs are flammable and pose fire hazards. Also, at least one substitute for CFCs is suspected of being toxic.

The fact that these effects were not fully evaluated does not necessarily mean that the regulations in question should not have been promulgated. The benefits of the regulations may still outweigh the costs. It was not possible to accurately assess the magnitude of these inter-media and substitution effects for all the regulations we reviewed. Information in addition to the background and support documents for the regulations is required. In some cases, these effects may be small, even negligible. In other cases, however, they will be important.

The case studies provide the opportunity to examine more closely the costs imposed by inter-media and substitution effects. To do so, the health risks associated with substitutes or the transference of a pollutant from one medium to another will have to be estimated. Information about the actual substitutes or pollution transference is required. This is not likely to be obtainable in all cases, thus, necessitating bold assumptions about the scope of the problem. In such cases, wide ranges of uncertain effects will be estimated.

Estimating the expected adverse health impacts of these effects will be handled in a manner similar to that used to estimate the reduction in risk resulting from the regulations. To the extent possible, existing information on exposed populations, exposure quantities, exposure durations, and dose/response relationships will be utilized. Assumptions or short-cuts will be required when information is lacking.

Regulatory uncertainty can be an important influence on estimates of these costs. For example, methylene chloride, a substitute for CFCs in several uses, has recently been added to EPA's list of toxic air pollutants. Future control of methylene chloride may reduce the adverse effects from controlling the use of CFCs. Generally, however, the uncertainty over future regulations is so great that it can not be incorporated into the analysis in a meaningful way. Although this can be examined further in the case studies, its examination is unlikely to prove very fruitful.

Balance of Trade

Although environmental regulations impose costs on domestic industry, the Department of Commerce has concluded in several studies that they have had very little or no effect on international trade.[10/] Compliance costs in the

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[10/] U.S. Department of Commerce. The Effects of Pollution Abatement on International Trade, III, April 1975 and IV, November 1976.

U.S. were found to be higher than those abroad, but other factors affecting balance of trade were found to be more important; such as demand-supply relationships, transportation costs, the existence of captive markets, and trade barriers. In specific instances, however, regulations may have significant effects.

To measure the effect a regulation has on balance of trade, the net change in balance of trade must be examined. For example, the industry that produces product A may lose production to imports because of the compliance costs of a regulation. However, if the foreign competitor that ships increasing amounts of product A into the U.S. also purchases increasing amounts of product B from U.S. companies, then the industry that produces product B in the U.S. increases exports. The net change in exports is the appropriate measure of the effect of the regulation. Additionally, if increased imports lower domestic prices, there may be a consumer surplus gain that should also be considered.

This net impact is difficult to estimate in practice because we only have a rough idea of what happens to the dollars we send overseas for imports. A different percentage of the money we spend on imports returns depending on the country to which it was "sent". If balance of trade impacts are large for a given regulation, then some "average" of the appropriate percentage of money respent in the U.S. could be used. The appropriate value for balance of trade costs is this approximate net change in production (minus any employment costs counted elsewhere).

As we have not found significant balance of trade effects due to the regulations we reviewed, we will not develop this method for valuing balance of trade effects any further. Should these effects be significant in regulations reviewed as part of any cost-effectiveness study to follow, we will more fully develop and apply an approach to this valuation at that time.

Employment

Pollution abatement requirements are often identified as a cause of unemployment. In the aggregate, however, environmental regulations have had a beneficial effect on total employment. In a 1978 macroeconomic study, DRI predicted that through the mid-1980s more jobs would be created by environmental regulations than would be lost or "not created". Beginning in the mid-1980s the effect is expected to become slightly negative.

Although the net effect of pollution abatement requirements on the total number of jobs may be small, the loss of jobs has been concentrated primarily in two regions, II and V, and the skills required for new jobs often do not correspond to the skills possessed by people losing jobs. Therefore, changes in employment patterns impose distributional costs. Also, in areas where a large number of people become unemployed simultaneously, entire communities may become economically depressed. This more extreme situation can occur if the primary employer in a town decides to close. The effect that environmental regulations can have on plant closures will be discussed separately below.

Assumptions about the general level of economic activity, costs and methods of compliance, and rates of return are needed to predict the number of jobs expected to be lost in an industry affected by a regulation. As mentioned previously, these factors cannot be estimated with a high degree of certainty. Therefore, the effects of individual regulations on employment are difficult to predict accurately.

In retrospect it is also difficult to measure exactly how many jobs have been lost due to a particular regulation. The cause and effect relationships between employment and other factors, such as the general level of economic activity, must be separated out. Again, the uncertainties in these relationships reduce the accuracy with which the employment effects of environmental regulations can be measured.

While it is difficult to predict precisely the number of jobs lost due to regulation, each economic impact statement we reviewed concluded that only a small number of workers would be left jobless as a result of the regulation. The economic impact statement for the Clean Air Act, Section 112 vinyl chloride regulation estimates only thirty employees industry-wide would lose their jobs. The economic impact analysis of the TSCA CFC regulation predicts that a small number of jobs would be lost in two segments of the aerosol industry; plants that produce aerosol spray cans and their components, and plants that fill the aerosol spray cans.

When jobs, such as these in the aerosol industry, are eliminated, the economy suffers a real loss equal to the marginal productivity of the newly unemployed individuals. In a perfectly competitive labor market, this is equal to the gross compensation paid to the workers. The economy only suffers this loss as long as the individuals remain unemployed. Therefore, to value the cost of lost jobs, the present discounted value of lost gross compensation over the time that individuals were unemployed might be calculated. (To prevent double counting, this should not be included if the total value of lost output is counted elsewhere.)

To be complete, this valuation of lost jobs should be balanced by the increase in jobs in other sectors of the economy caused by the regulations. For example, the vinyl chloride regulation may create jobs in the pollution control equipment industry. Also, the CFC regulation probably "transfers" jobs from the aerosol industry to the mechanical-pump-spray industry. When this balancing is done, as the DRI results mentioned above indicate, in the aggregate the total number of jobs remains about the same.

There was not sufficient information about the regulations we reviewed to determine if the employment effects of individual regulations also balance out. This is likely to vary considerably from regulation to regulation. Some inferences about the net employment effects of a regulation may be drawn from information on the additional operation and maintenance requirements imposed by that regulation. If these appear to be of the same magnitude as the predicted reduction in jobs, then the number of jobs may not change much as a result of that regulation. In the absence of any information about a regulation's net effects on employment, the value of lost employment as measured above should be included in an estimate of the upper bound of the costs of that regulation.

Even when the net employment effects of a regulation are small, its imposition of distributional costs caused by changes in employment patterns may be important. To value these costs, estimates of the search and relocation costs incurred by individuals changing jobs due to a regulation are required. These distributional costs are likely to be very small for the regulations we examined because the number of jobs affected is very small. However, if the total number of jobs affected is large, this cost may be significant.

Plant Closures, Effects on Small Businesses, and Changes in the Competitive Structure of Industries

The effects of regulation on plant closure, competitiveness and small businesses are very closely related and are usually addressed together in the economic impact statements. The impact statements we reviewed indicate that small businesses are more often significantly affected by environmental regulation because of their lower accessibility to capital necessary for needed control technologies or conversion to new or other processes. Plant closures, whether they be large or small plants, can affect the competitive structure of an industry. A fairly competitive industry may become oligopolistic with uncertain impacts.

All of the economic impact statements we reviewed discuss the possibility of smaller businesses being unable to manage the increased cost and closing. The actual number of closings is predicted only for the vinyl chloride industry with four small polyvinyl chloride production plants predicted to close.

The CFC economic impact statement estimates a larger impact on small firms because of small firms' more limited access to the financing for capital conversion and R&D costs. For example, conversion costs for the filler industry (firms involved in filling aerosol containers) tend to be prohibitive for small fillers, so an adjustment in market share favoring the large fillers appears probable. Larger organizations have more influence on suppliers and may be able to obtain necessary new equipment faster than small firms, even when the latter have the access to capital.[11] New equipment is necessary for both small and large firms. A larger firm can amortize the new equipment costs over several filling lines, making the cost relatively less than to small firms, which can amortize the cost only over a smaller number of lines. It is likely that these conversion costs could become prohibitive to some smaller filler firms, and they could go out of business.

The prediction of potential closure of plants is based upon a projection of the increased cost to plants as a result of each regulation. It is difficult to predict how many plants will close without an estimate of changes in demand and market conditions. It is clear that the CFC and vinyl chloride regulations will at the very least contribute to several plant closures.

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[11] International Research Technology Corporation and Policy Models, Inc., The Economic Impact of Potential Regulation of Chloroflurocarbon - Propelled Aerosols, (Washington, D.C., EPA, 1977), p. 4-22.

It is difficult to value the loss of a plant. If the closure does not affect the structure of the industry or demand for raw materials, the monetary cost is equal to the value added that firm used to produce. This includes the value of both the labor and capital inputs that the firm used. Over time, the labor will become reemployed and the capital if fungible, will be switched to another productive use. Therefore, the appropriate figure would be the present discounted value of the lost value added over time. Changes in value added for industries are measured by the Commerce Department. Attributing these changes to individual environmental regulations is problematic, however. Therefore, information about the individual plants affected is required to estimate this cost.

In addition, plant closures have effects on competitiveness and small businesses. Competitiveness effects may be measured as changes in the price of a good as a result of firms going out of business. The loss of small businesses is also generally viewed with disfavor for non-economic reasons. Whether based in the spirit of protecting the small entrepreneur exercising freedom, or in the need to reduce barriers to entry to maintain a competitive economic system, the value of protecting small businesses from government induced costs is recognized. It is difficult to monetize these values, however, and for our purposes, the costs of small businesses going out of business will be measured the same way as plant closures, the present value of the lost value added of that business.

Our valuation of plant closures does not reflect the extreme situations that can be caused when the primary employer of a town closes down. The economic impact statements we reviewed do not indicate that any one town would face economic collapse due to environmental regulation. If, in the case studies, a situation arises where a plant closure caused by a regulation results in an extreme situation, an approach to valuing the costs will be developed. Some economic multiplier used in conjunction with total lost wages will probably be at least one potential alternative for investigation.

Price Increases

The effect that environmental regulations have on relative prices varies from industry to industry and regulation to regulation. In the aggregate, environmental regulations are expected to have little effect on domestic prices. In a 1978 macroeconomic study, Data Resources, Incorporated (DRI) estimated that environmental regulations would add approximately 0.3 percent per year to the increase in wholesale prices from 1970-1986. The impact on individual industries may vary, however. For example, Arthur D. Little Incorporated (ADL) predicted in a report prepared for EPA that environmental regulation could cause copper prices to increase by over twenty-three percent by 1983 and by thirty percent by 1987.

Price effects are difficult to measure. To predict price effects, assumptions about future demand, capital availability, and general economic activity are required. The necessity of an industry to expand to meet future demand will affect price predictions. Actual price effects are also difficult to measure in retrospect because it is difficult to separate the effects of general economic activity from those of the regulation. The costs and benefits of price effects have distributional consequences but may have small net impacts. Within an industry, plants with low control costs will benefit and those with high control costs will suffer. This may produce regional distributional effects. Also, the change in relative prices will affect the composition of GNP.

Environmental regulations can cause prices to increase in several ways. Capital, operating, and maintenance costs of compliance may change the price at which the industry can supply goods. Also, if small firms are forced to close due to changes in economies of scale caused by control technologies, the industry may become more oligopolistic. Depending on elasticities of demand and substitution, an increase in industry concentration can result in an increase in prices. Prices may also increase to reflect increases in unit labor costs caused by slowed growth in productivity or reduced investment in new productive capacity. These various individual effects are discussed elsewhere in this report.

The background documents for all of the regulations we examined discussed the changes in prices that could result from compliance. The economic impact analysis of the proposed volatile organic compound (VOC) regulation (Clean Air Act, Section 111) predicted, at most, very modest price increases in the synthetic organic chemical manufacturing industry. Most costs of compliance were expected to be offset by the value of recovered product. The vinyl chloride regulation (Clean Air Act, Section 112) was predicted to increase prices of polyvinyl chloride (PVC) resins by an average of 7.3 percent which would increase consumer prices of products containing PVC resin by 3.5 percent. This increase in consumer prices was not believed to be large enough to bring about substitution for PVC resin products.

Negligible increases in the price of drinking water were predicted as a result of the trihalomethane (THM) regulation (Safe Drinking Water Act). Similarly, the chlorofluorocarbon (CFC) aerosol propellant ban (TSCA) was not expected to increase consumer prices. The TSCA PCB regulation, however, was expected to have varying price impacts. The price of capacitors was predicted to increase by as much as ten to twenty percent, and the price of some electromagnets that used PCBs was predicted to increase even more. For most other products, the price effects of the PCB regulation were anticipated to be very small.

These price increases described above can have two secondary effects which impose real costs on the economy. First, price increases add to the rise of the general price level, i.e., inflation. A real redistribution of wealth occurs because some sectors tend to be creditors and others debtors. A real loss is realized because the capital markets adjust imperfectly. Also, it becomes more costly to hold currency, and individuals incur additional costs maintaining their assets in interest bearing investments.

A single regulation is not likely to affect inflation significantly. As mentioned above, DRI forecasted that all environmental regulations will add only 0.3 percent to annual increases in wholesale prices from 1970-1986. Therefore, the costs imposed by price effects through their influence on inflation need not be considered in the cost effectiveness of individual regulations.

Second, price increases can impose real costs if resulting changes in relative prices are not anticipated. The value of existing capital equipment that uses a product whose price has increased can decrease dramatically due to the unexpected price shift. Much of the revaluing of capital stock may be a transfer between owners of different types of equipment, but a real cost is imposed because for a time, capital is not invested in its most highly valued uses. When price changes are large and unanticipated, this cost can be large. Estimating its magnitude, however, is problematic.

Of the regulations we examined, only the PCB rule appeared to potentially impose this cost. The price of some items, e.g. electromagnets, may increase substantially due to the ban. From the information reviewed, it was not possible to assess the extent to which this may affect the value of existing capital equipment. To do so, the uses of electromagnets would have to be investigated.

Changes in relative prices should be examined closely in the case studies. When changes are small, the costs outlined here may be ignored. However, if the price shift is large and the product is an intermediate good, then this cost should be estimated by examining the effect the price change has on the users of the product. Information in addition to the support documents for the regulations would be required to determine the magnitude of these costs. The incremental costs that users of the intermediate products bear as a result of unanticipated changes in relative prices should be estimated.

Since, for the regulations we examined, price effects were not major, a method for valuing their impacts has not been finalized. If unanticipated changes in prices are large in one of the case studies, this topic will be investigated further.

Reductions in Productivity and Investment in New Productive Capacity

Insofar as pollution abatement investments displace other more productive business fixed investments, the rate of increase in the productivity of labor may be reduced. In a 1978 macroeconomic study, DRI predicted that productivity would be 1.4 percent lower in 1986 than it would be without pollution abatement requirements. One of the underlying assumptions of this forecast is that pollution control equipment contributes nothing to productive capacity. This assumption, as DRI acknowledges in its report, does not always hold true. Therefore, this 1.4 percent estimate may be slightly high.

Environmental regulations can also increase the cost of new capacity. This may discourage pre-emptive investments in capacity built in anticipation of demand. In addition, however, the cyclical sensitivity of older capital stock is reduced, and there is greater opportunity for competitors to bring less capital-intensive production facilities on-line quickly while supply lags behind demand. These two latter effects, reduced cyclical sensitivity and greater opportunity for less capital-intensive production facilities, will actually benefit segments of the affected industry that are in positions to take advantage of the situation.

These effects are generally not measured separately as costs of regulations. To prevent double counting of costs, assumptions concerning these effects may be incorporated into predictions of price effects. Changes in the (labor) productivity of an industry are measured by the Commerce Department, but it is difficult to attribute the cause and effect of environmental regulations to these figures. Some indepth analysis of individual industries have tried to measure the productivity effects of regulation. Even once they were measured, however, they were not valued.

Effects on productivity and investment in new productive capacity were discussed very little in the background documents of the regulations we examined. Three regulations (proposed volatile organic chemicals (VOC); vinyl chloride (VC); and aldrin/dieldrin) were suspected of possibly having these effects. Only the VOC economic impact analysis attempted to quantify the effects by examining how changes in productivity could affect prices. In addition, the PCB regulation (TSCA) could potentially affect the productivity and investment patterns of some industries, but there was insufficient information to assess to what extent this is the case.

The fact that this issue was only briefly addressed in the background documents of several regulations is indicative of what appears to be its small magnitude. To handle this issue in the case studies, the expected capital investment required by the regulations should be compared with the total capital expenditures expected for the appropriate segments of the affected industries. If it is relatively small, then these productivity and new capacity effects can probably be ignored. If a relatively large portion of expected capital investment is required for compliance, then this issue should be examined closely.

For those regulations for which this issue is significant, the effects of productivity and investment changes on prices should be assessed. This was done for the proposed VOC regulation under the Clean Air Act by estimating the change in unit labor costs. The approach described in the previous section should be used to value changes in prices. In the absence of an industry specific supply/demand model, however, more sophisticated analyses of these effects is not possible. Therefore, some portion of these impacts may have to go unmeasured in the case studies.

Capital Availability Problems

Pollution abatement may require large capital investments. However, concern that environmental regulations could themselves cause very tight capital markets does not appear to be justified. In a 1978 macroeconomic study, DRI predicted that environmental regulations would have a small effect on interest rates causing real fixed business investment to be reduced by 2.5 percent and housing investment to be reduced by 4.7 precent by 1986. The impacts during 1973 and 1977/78 accounted for most of the predicted reduced investment because near credit-crunch conditions existed. Reduced financing pressures caused by removing the pollution-related expenditures could have had a major impact on stimulating new investment.

This general transfer of investment capital from housing to pollution abatement equipment is very difficult to value, even if it could be measured for individual regulations. It is unlikely that individual regulations could require sufficient investment to affect capital markets. This was borne out in our examination of six regulations. The capital requirements were negligible relative to the capital market as a whole. Therefore, assigning a portion of this macroeconomic effect to individual regulations would not be very meaningful.

Individual regulations can affect the ability of specific firms to raise needed capital. This was discussed in the previous section on productivity and investment in new productive capacity.

Effects on Innovation

Environmental regulations can affect the allocation of research funds. Some regulations may be "technology forcing", thus requiring, for compliance, research on new technologies or on the application of existing technologies. Also, a regulation may make research involving a substance, such as chlorofluorocarbons (CFCs) much less attractive because of the likelihood of future control.

None of the support documents we reviewed in our examination of the regulations assessed effects on research. In relation to the control of CFCs, the National Academy of Sciences, NASA, and other groups have expended considerable resources exploring the relationships between CFC emissions, ozone depletion, and the effects of potentially increased exposure to ultraviolet rays. It would be inappropriate, however, to include the cost of this or similar research as a cost of the CFC regulation.

The effect of an individual regulation on the allocation of research funds cannot be determined. Even if we could estimate the effect that regulations do have, for example in the aggregate, we would strain even to value these effects as positive or negative. Therefore, the effects of environmental regulations on innovation cannot be considered further.

Innovation, however, can affect the cost of pollution abatement. New technologies are often adopted that reduce the cost of compliance. These changes in costs would be reflected in actual compliance cost data which are discussed elsewhere in this report.

3.3 SUMMARY

This chapter outlines the feasibility of estimating, for purposes of comparison, the total costs of existing and proposed environmental regulations. "Total costs" means the net impact a regulation has on the welfare of society exclusive of the risk reduction effects discussed in the previous chapter. Total costs include changes in consumers' and producers' surplus caused by the direct costs of compliance; transactions costs imposed by unanticipated changes in employment patterns, investment schedules, relative prices, and the competitive structure of industries; and costs imposed by the transference of pollutants from one medium to another or by the substitution of one substance for another. The following three issues were examined for several regulations to assess the feasibility of estimating these costs:

1. Quality of cost estimates;
2. Joint costs, common costs, and economies of scale; and
3. Nonmonetary compliance costs.

Exhibit 3-1 highlights the results of the analysis. The most significant costs of the regulations were found to be the capital and operating costs. In most cases, these costs were adequately identified and estimated in the economic analyses of the regulations we reviewed.[12] The quality of these estimates varies, however. In several cases, the assumptions upon which the analyses were based were not adequately identified or documented. The quality of the cost estimates remains uncertain for these. On the whole, estimates of capital and operating costs were found to be usable, although some of the assumptions used in the analyses should be documented to increase the reliability of the estimates.

The analysis of the chlorofluorocarbon (CFC) aerosol propellant ban estimated the loss in value added in the industries adversely affected by the regulation. Change in value added is not a measure of change in welfare for society, although it indicates the magnitude of the impacts on the affected industries. Value-added losses in one industry may be offset by increases in value added in other industries. The exclusion from the analysis of the industries likely to benefit from the CFC ban is a major obstacle that must be overcome to assess the net economic impacts of the regulation.

Joint costs were not addressed in the regulations reviewed. All costs incurred are assumed to be the result solely of the individual pollutants under analysis. This assumption appears to be valid in most cases, and does not bias the estimates of the costs of the regulations. In situations, where joint costs could be a problem, e.g., the trihalomethane regulation promulgated under the Safe Drinking Water Act, there was insufficient information available to assess its magnitude.

The nonmonetary compliance costs of the regulations reviewed were very small and in some cases, negligible. Despite the general insignificance of these nonmonetary costs, there is potential for costs to be imposed when a pollutant is transferred from one medium to another without adequate control or when the substance that is substituted for a regulated substance poses hazards. These "inter-media and substitution effects" were found in four of

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[12] Cost analysis is not required for National Ambient Air Quality Standards (NAAQS) promulgated under Section 108 of the Clean Air Act. Cost estimates may not be available for many NAAQS regulations.

EXHIBIT 3-1

HIGHLIGHTS OF THE RESULTS OF THE COST ANALYSIS

Regulations Reviewed	Costs Estimated	Quality of Cost Estimates	Joint Costs, Common Costs and Economies of Scale	Nonmonetary Compliance Costs
Aldrin/Dieldrin, Endrin, DDT, Toxaphene; FWPCA.	Capital and operating costs.	Uncertain; all assumptions not explicitly identified.	Not addressed.	None identified.
Trihalomethane; SDWA.	Capital and operating costs.	Approach seems thorough but assumptions are not explicitly identified.	Not addressed: these may be important for this regulation.	Intermedia transfer of pollutant and substitution affects.
PCBs; TSCA.	"Transitional" decontamination costs and capital and operating costs of the restrictions.	Appears to be good; all assumptions explicitly shown.	Not addressed.	Substitution effects; Potential price effects.
CFCs; TSCA.	Loss in value added in the affected industries.	Uncertain; data not well documented.	Not addressed.	Substitution effects; Small employment effects.
Volatile Organic Compounds, CAA (proposed).	Capital and operating costs.	Appears to be good, but questions are raised.	Not addressed.	None identified.
Vinyl Chloride, CAA.	Capital and operating costs.	Appears to be very good.	Not addressed.	Intermedia transfer of pollutant; Small employment effect.
Benzene, CAA (proposed).	Capital and operating costs, employment and balance of trade effects.	Capital and operating costs appear to be very good; models used to predict other costs are less certain.	Not addressed.	(Not examined.)

six regulations examined for nonmonetary costs, but the precise magnitude of these costs could not be identified from the available data. Insofar as these costs can be estimated using the techniques identified in the previous chapter to measure risk, the presence of these costs should not impede the estimation of the total costs of regulations.

Therefore, on the whole, the regulations examined indicate that the costs of regulations can generally be estimated. There are, of course, many uncertainties in the accuracy of the estimates. The uncertainties can be reduced by re-analyzing portions of existing economic impact estimates. In some cases, data collection may be required.

4. CASE STUDIES

An important part of this feasibility study is the development of several cost-effectiveness analyses of EPA toxics regulations. These analyses or case studies demonstrate how many of the benefit and cost issues discussed earlier in this report can be addressed in specific situations. The case studies also helped us identify the benefit and cost issues to study.

We developed a list of ten EPA toxics regulations as candidates for case studies. These are shown in Exhibit 4-1. For each regulation we developed a rationale for its inclusion as a candidate. Generally, a regulation was included on this list for one of two reasons. First, it may have been thought to be representative of the regulations developed by a particular EPA program office. Second, it may have been considered to offer a good opportunity to study specific aspects of the cost and effectiveness issues we had identified as important to resolve in order to compare EPA regulations on a cost-effectiveness basis.

Of the ten candidate regulations, we focused attention on the following five:

1. Lead in ambient air (Clean Air Act, Section 109)
2. Lead (and lead compounds) spilled into navigable waters (Federal Water Pollution Control Act, Section 311)
3. Lead in drinking water (Safe Drinking Water Act)
4. Vinyl Chloride emissions (Clean Air Act, Section 112)
5. Chlorofluorocarbons used as aerosol propellants (Toxic Substances Control Act).

The case studies themselves are in Appendices A,B, and C. The three lead cases are in Appendix A; having been put together to facilitate comparisons between them.

The five case studies illustrate well the differences between regulations in terms of analytical treatment required in order to include them in a cost-effectiveness comparison. Also, the limitations of such comparisons are well illustrated. The Federal Water Pollution Control Act (FWPCA), Section 311 lead regulation and the Safe Drinking Water Act (SDWA) lead regulation are alike in not having either an EPA cost analysis or benefit analysis. We demonstrate how one might structure an analysis of the benefits and costs of the SDWA lead regulation. Similarly, we demonstrate how such an analysis for the Section 311 lead regulation is probably infeasible.

The lead cases clearly show what types of cost-effectiveness comparisons between them and with other non-lead regulations can be made. This

EXHIBIT 4-1

CANDIDATE CASE STUDY REGULATIONS

Statute		Regulation
Toxic Substances Control Act		Chlorofluorocarbons
Clean Air Act	Section 112(NESHAPS)	Vinyl Chloride
	Section 108-109(NAAQS)	Lead
Federal Water Pollution Control Act	Section 307	Aldrin/Dieldrin
		Polychlorinated biphenyls
	Section 311	Aldrin
		Lead
Federal Insecticide, Fungicide and Rodenticide Act		Aldrin/Dieldrin
		2,4,5 T
Safe Drinking Water Act		Lead

demonstration has important implications to the design of a full-scale cost-effectiveness study of EPA regulations.

Both the vinyl chloride and the chlorofluorocarbons cases had EPA cost estimates supporting them. In both cases, we made modifications to the estimates to improve their accuracy.

The vinyl chloride regulation is one for which EPA had developed an analysis of benefits. This simplified our task, but we still had to make some modifications to the EPA analysis. We also discuss in the case additional analytical tasks necessary to make the resulting cost-effectiveness numbers derived comparable to other than carcinogen regulations.

The main purpose of the cases was to provide an opportunity to study the benefit and cost issues and to develop a methodology to do further such studies. As such, the emphasis was not on arriving at final, fully defendable cost-effectiveness numbers. In some instances, for purposes of illustration, some assumptions are made that would not have been made in a full-scale cost-effectiveness analysis. For this reason, any numerical "results" developed in these cases should be reviewed with this in mind and should not be used for purposes beyond the scope of this feasibility study.

5. CONCLUSIONS

This study has sought to determine the feasibility of developing and comparing cost-effectiveness measures for EPA toxics regulations. As we have stated, for that to be feasible, three conditions must be met:

1. data needed to perfom the analysis must be either available or possible to develop;

2. potential methodological difficulties must be resolvable; and

3. the work must be able to be accomplished within reasonable budgetary constraints.

We discuss each of these conditions in turn.

5.1 DATA

The type of data needed for a cost-effectiveness analysis of toxics regulations has been reviewed extensively in the discussion of benefit and cost issues and has been illustrated in the case studies. Regarding cost data, capital and operating costs are most important. In most cases, we found these costs to be adequately identified and estimated in the economic analyses of the regulations we reviewed. We found some of the assumptions used in the economic analyses to need better documentation, without which the reliability and accuracy of cost estimates cannot be determined. There are, of course, many uncertainties in the accuracy of the estimates. The uncertainties can be reduced by re-analyzing portions of existing economic impact estimates. In some cases, data collection may be required.

The case studies illustrate well the range of cost data availability. Very little cost data exist for the Safe Drinking Water Act (SDWA) and the Federal Water Pollution Control Act (FWPCA), Section 311 lead standards. Information is sufficiently lacking so as to make it very difficult to estimate costs for the FWPCA standard. Cost estimates for the SDWA standard would also be difficult to develop, but it could be done as we have outlined. Some cost data are available for the Clean Air Act, Section 109 lead standard. However, that data would need to be updated before it could be used to develop reliable cost estimates.

Thorough cost analyses were available for both the Clean Air Act, Section 112 vinyl chloride standard and the Toxic Substances Control Act chlorofluorocarbons standard. However, it is important to note that even in these cases, we made several changes to improve the estimates. Also, the difficulties involved in using the value-added approach, as was done for chlorofluorocarbons, was reviewed in Chapter 3.

Regarding benefit data, we found that for only a few regulations were such data developed by EPA. For most regulations, the data were presented to show the seriousness of an exposure problem, but not to show in a quantitative way what the regulation would accomplish in a specific period of time. For most regulations, data showing the risk and the affected population is either available or can be developed. The availability of data for benefits analysis is summarized in Exhibit 2-2 of Chapter 2.

The cases studies provided a full range of benefits data availability. The SDWA and the FWPCA lead standards had very little, readily available benefit data. Although we were able to develop benefits data for the SDWA standard, meaningful benefits data for the FWPCA lead standard could not be developed. Data available for the air lead standard, while extensive, focused on the health effects of lead, and did not provide the other information necessary to assess the level of risk reduction brought about by the standard. Risk reduction estimates could be developed with a significant effort.

The vinyl chloride emissions standard was one of the few for which a risk reduction analysis had been accomplished. However, that analysis needs to be updated with new methods recommended by the EPA Carcinogen Assessment Group since the vinyl chloride standard was promulgated.

The TSCA CFC regulation also had benefits data. We elaborate on the EPA developed data and present a fuller range of possible benefits in the case study.

In summary, it can be seen that the availability of needed cost and benefits data varies widely across regulations. We have demonstrated in several of the cases what is involved in filling some of these data gaps. Based on our review of the available data and our experience with the cases, we conclude that data availability does not represent a major barrier to accomplishment of the envisioned cost-effectiveness study.

5.2 METHODOLOGICAL DIFFICULTIES

Potential methodological difficulties (issues) were reviewed in Chapters 2 and 3 as well as in the cases. Regarding cost issues, we found the quality of cost estimates for regulations to vary. On the whole, however, estimates of capital and operating costs were found to be usable.

The review of the issue of valuing and comparing nonmonetary compliance cost (e.g., employment effects) indicated these costs to generally not be of major significance. Estimates of the costs imposed by nonmonetary effects are generally based on estimates of expected market reactions to capital and operating costs. Methods are proposed for estimating the costs imposed by several of the nonmonetary impacts found to be potentially significant. Other such methods can be developed when they are found to be needed in the analysis of a particular regulation.

For the benefit issues, methodological difficulties are of more concern. One such difficulty involves expressing the benefits of different regulations in comparable units. This cannot always be done. The lead case studies illustrate this point well. Since there are no quantitative measures of the degree to which the lead regulations actually reduce the incidence of disease, comparisons of these regulations with other regulations along the commonly-used dimensions of health impact (e.g., deaths prevented, life-years saved) are not possible.

While comparisons cannot be made between the lead regulations and other non-lead regulations, comparisons can be made among different lead regulations. Direct comparisons of changes in the population at risk of the various lead-induced diseases can be made. This approach can theoretically be applied to any regulation (e.g., the Section 211 regulation limiting lead levels in gasoline) which effects lead exposure in the general population. Comparisons of this type have important implications from a toxics integration perspective.

The vinyl chloride case results in benefits expressed in numbers of cancers prevented. In this form, comparisons can be made with other regulations controlling carcinogens. Numbers of cancers prevented can be converted to life-years saved. With this benefit measure, the benefits of the vinyl chloride regulation can also be compared to the benefits of other non-carcinogen, life-years saving regulations.

Some regulations result in a variety of benefits. For example, the CFC aerosol propellant regulation is expected to prevent cases of both melanoma and nonmelanoma skin cancers in the United States. In the case study, we developed estimates for each of these types of skin cancer. We did not develop a common unit of benefit measure for these different cancers, thereby making comparisons of the cost-effectiveness of this regulation to other regulations difficult. We could compare the number of cancer deaths expected due to melanoma skin cancer with other regulations. This cancer death measure can be thought of as a lower bound on total benefits. Another approach would be to use reduced future medical costs as a lower bound of the value of preventing cases of nonfatal melanoma skin cancer and nonmelanoma skin cancer. The estimate of those two benefits could be subtracted from the total cost and the remaining cost estimate could be used to develop a cost-effectiveness measure per premature death prevented. Additional work is needed to improve the analytical methodologies available to commonly quantify multiple benefits.

Another methodological difficulty involves the nonmonetary benefits discounting issue. This was discussed extensively in Chapter 2 and its impact on benefits projections was illustrated in the chlorofluorocarbons case study. Currently, there is no concensus on the question of whether nonmonetary benefits should be discounted and what discount rate should be used.

In summary, there are a number of methodological difficulties in performing a cost-effectiveness analysis of EPA toxics regulations. However, these difficulties, for the most part, act to limit the types of comparisons that can be made between regulations rather than make the calculation of cost and benefits infeasible. Even with the limitations, many interesting and worthwhile comparisons can still be made. We conclude that a comparison of the cost and effectiveness of many regulations along a number of meaningful dimensions is not prevented by methodological difficulties.

5.3 BUDGETARY CONSTRAINTS

The budget required for a study of the cost-effectiveness of EPA toxics regulations is to a large extent a function of the number of regulations studied and the amount of needed data not readily available for the regulations selected to study. There is no fixed number of regulations that need to be included in the study. That is, more can most likely be learned, for example, by studying ten regulations than by studying eight regulations; however, a study of eight regulations would be worthwhile.

The amount of needed data not readily available can, of course, be controlled by the selection of regulations to study. The trade-off here is including some regulations important to study but which may have large data gaps against the cost of filling those gaps.

It should be kept in mind that the quality of the estimates of cost and benefits for regulations are, to a certain extent, a function of the amount of resources expended. Greater resource expenditure generally means that analyses can be constructed in greater detail, that more precise calculations can be substituted for rough approximations, and that data can be gathered to fill important gaps in knowledge. One of the major challenges in conducting analyses of this type is to determine the level of resources at which the results are sufficiently precise and reliable for their planned uses, but where resources have not been spent on needless elaboration or speculation.

In summary, the amount of information to be gained from a cost-effectiveness analysis of EPA toxics regulations is, of course, a function of the available budget; however, there is no high budget threshold required to be met before something useful can be done. Therefore, the cost-effectiveness analysis is feasible within reasonable budgetary constraints.

5.4 SUMMARY

We believe a cost-effectiveness analysis of EPA toxics regulations is feasible and can produce interesting, useful results. Feasibility has been determined through analysis of the three conditions for feasibility listed at the beginning of this chapter.

While the cost-effectiveness analysis is feasible, the task of making cost-effectiveness comparisons across regulations is one which must be addressed with a good deal of caution. Seemingly obvious conclusions may not be so obvious once the procedures used to make the cost and effectiveness estimates are compared and once the underlying constraints faced by EPA are reviewed. Concluding that this type of analysis is feasible, does not mean it will be easy. On the contrary, our feasibility study indicates that a substantial amount of data collection and analysis will usually be required to develop the necessary cost and benefit estimates. The approach taken must utilize existing data and accepted assumptions as much as possible.

APPENDIX A

CASE STUDIES OF THREE EPA LEAD REGULATIONS

APPENDIX A

CASE STUDIES OF THREE EPA LEAD REGULATIONS

A.1 INTRODUCTION

These case studies analyze and discuss three regulations promulgated by the Environmental Protection Agency which have as their goal the reduction of lead contamination of the environment, and the reduction of human exposure to lead from air and water. The regulations chosen for analysis are the National Ambient Air Quality Standard for Lead[1]; the Primary Drinking Water Standard for Lead[2]; and the Hazardous Substances Spill Reporting and Control Regulations Promulgated under Section 311 of the Federal Water Pollution Control Act, as applied to spills involving lead compounds.[3]

The National Ambient Air Quality Standard for lead, promulgated in October 1978, sets a maximum allowable level for suspended lead in air of 1.5 ug/M^3. The standard applies uniformly to all areas of the United States, and is implemented through the revision of State Implementation Plans to include controls of lead emission from stationary sources, such as primary and secondary lead smelters, copper and zinc smelters and grey iron foundries.

Another standard promulgated by EPA governing the amount of lead allowed in gasoline will also affect ambient air lead levels, especially in urban areas where traffic density is high. For this reason, evaluation of the effects of the ambient air standard must also take into account the effects of the gasoline lead standard.

The rationale for the selection of the 1.5 ug/M^3 ambient limit was based solely on public health grounds; EPA determined that the chosen level was necessary to prevent sizable numbers of children (the most susceptible population to lead toxicity) from developing dangerously elevated blood lead levels. Section 109 of the Clean Air Act does not allow EPA to take economic factors into account in setting National Ambient Air Quality Standards. Even so, EPA did conduct some cost analyses, and in addition, the affected industries developed prospective estimates of the economic impacts of the standard. No retrospective data on the costs or effectiveness of the regulation are available at this time.

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[1] 43 Federal Register 46246, Oct. 5, 1978.

[2] 40 Federal Register 59565, December 24 1975, amended by 45 Federal Register, 57342, August 27, 1980.

[3] 43 Federal Register 10479, March 13, 1978, 44 Federal Register 50776, August 29, 1979, (both as amended).

The Primary Interim Drinking Water Standard for lead was promulgated in December, 1975, along with similar standards limiting the concentration of metals that would be allowed in finished drinking water. The standard which limits the maximum allowable concentration of lead to 50 ug/l (0.05 mg/l), is identical to a Public Health Service standard set in 1962. It does not appear that any quantitative analysis of the benefits of this standard have been performed, and even though a National Academy of Sciences report published in 1977 suggested that the standard might not be protective enough, the standard has not been revised.

No study of the costs of the regulation were undertaken at the time of promulgation. The principal source of lead in drinking water is the use of lead distribution pipes. Thus, the major costs would be those borne by municipalities and the federal government in replacing these pipes. Little data is available as to the rate at which such pipes are in fact being replaced.

The Hazardous Substances Regulations promulgated under section 311 of the Federal Water Pollution Control Act designate certain chemical substances to be "hazardous" on the basis of their toxicity and other properties. Each hazardous substance is then assigned a toxicity category, based on its degree of toxicity, and "reportable quantities" of each chemical are established. Every spill of a substance into navigable waters which exceeds the reportable quantity must be reported to the Coast Guard, so that cleanup action can be initiated. Spillers are also assessed a fine which is calculated according to the amount of chemical spilled, and its toxicity and persistance in the environment. Several hundred compounds, including eleven inorganic compounds of lead, and one organic compound (tetraethyl lead) are classified as hazardous substances.

The rationale for setting the standard and fine schedule was presumably the prevention of harm to human health and the environment. No quantitative assessments of the benefits of the regulations were undertaken, however. Several studies were commissioned to estimate the economic impacts of the 311 regulations, although none specifically addressed the problem of lead spills.

In the sections that follow, the feasibility of developing estimates of the costs and effectiveness of each regulation is discussed. Section A.2 discusses the issues involved in developing cost and effectiveness measures for the ambient air quality standard. A sample cost estimation is undertaken to illustrate several important analytical issues. Section A.3 discusses the drinking water standard, and included in this section is an illustration of the way in which estimates of the effectiveness of the standard in reducing the advance effects of lead exposure can be developed. Section A.4 discusses the Hazardous Substance regulation, where a lack of data does not allow the development of quantitative estimates of either costs or effectiveness. Section A.5 summarizes the results of the analyses of the three regulations and discusses the implications for the comparison of the regulations' costs and effectiveness, with each other, and with other environmental regulations.

A.2 NATIONAL AMBIENT AIR QUALITY STANDARD FOR LEAD

A.2.1 Effectiveness of the Air Lead Standard

a. Existing Effectiveness Studies and Risk Assessments

Section 109(b)(1) of the Clean Air Act requires the Environmental Protection Agency to set primary ambient air quality standards in such a way as to "protect the public health . . . allowing an adequate margin of safety." In the preamble to the lead standard, EPA describes how the standard was set to assure that only a very small fraction of all children (the most sensitive population) would experience blood levels about 30 ug/dl, which was determined to be the maximum "safe" level.[1] This analysis, based on the data presented in the Criteria Document,[2] provides a useful basis for risk assessment, but does not, in itself allow quantitiative estimates of risk reduction of effectiveness to be developed. While arguments have been raised as to various assumptions made by EPA in their judgments about what constitutes a "safe" level and why, no one to our knowledge has undertaken a detailed analysis of the quantitative reduction in risk that would be produced by the lead standard. As will be discussed below, the reason for this is that the relationships between emission levels and ambient exposure, between exposure and changes in blood lead levels (or other physiological measures of lead absorption), and between blood lead levels and the incidence of clinical symptoms are very complex, and difficult to analyze quantitatively.

b. Feasibility of Conducting Effectiveness Analysis and Risk Assessment

Based upon an analysis and review of the recent literature and the available data about the environmental effects of airborne lead, we believe that a useful quantitative analysis of the effectiveness of the lead standard could be performed, and would enable meaningful comparisons to be made between it and other regulations intended to reduce population exposure to lead (such as the Primary Drinking Water Standard analyzed below). It is probable, however, that such an analysis would not include a quantitative estimate of the number of persons suffering from a specific lead-induced disease state. Rather, the two regulations could be compared on the basis of the total reduction in population exposure to lead, or on the basis of the total amount of lead actually absorbed by exposed populations. In addition, it is possible that estimates could be developed of the changes in the blood lead levels of the exposed population brought about by the air lead regulation and in the number of persons whose blood lead levels would be above certain "threshold" levels, at which they are known to be at risk of suffering the various toxic effects of lead. Thus, estimates of the total population at risk for each of these toxic effects could be developed, and the reductions in these populations brought about by the regulation could be compared. Such an analysis has previously been undertaken in support of the OSHA standard limiting airborne lead exposure in the workplace.[3]

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[1] 43 Federal Register 46246-46261, October 5, 1978.

[2] Air Quality Criteria Document for Lead, Office of Research and Development, U.S. Environmental Protection Agency, December 1977.

There are, however, a number of important practical and theoretical issues that would need to be addressed, if such an analysis were to be performed.

c. Important Analytical Issues in Effectiveness Analysis

1. Joint Benefits with the EPA Standard Limiting the Lead Content in Gasoline

Estimating the benefits of the Ambient Air Quality Standard for Lead is not simply a matter of assuming that the lead standard will have the effect of reducing the average lead levels in all the areas of the country now above 1.5 ug/M^3 to that level. If that were so, effectiveness estimation would be much simpler.

Unfortunately, a major determinant of lead levels in urban air is the amount of lead contained in gasoline. EPA regulations promulgated under Section 211 of the Clean Air Act require gradual but significant reductions in the amount of lead allowed in gasoline. Thus, all of the changes in urban air lead levels (there have been significant reductions in recent years) cannot be attributed to the ambient air lead standard, nor can simple projections of this data be used to predict future reductions in lead levels brought about by the ambient standard. Since, in effect, the ambient lead standard is actually a stationary source standard, it would be necessary to calculate for all sources the actual reduction in emission which would be required, and the actual reduction in exposure which would occur as a result. It would be necessary to take into account the decreasing background levels of airborne lead that would occur as a result of the gasoline lead regulation. The decreasing background lead levels would affect not only the extent of changes in exposure from emission reductions that would be required of individual stationary sources,[4] but would also affect the background blood lead levels which would determine, along with stationary source exposure, the effectiveness of the regulation in reducing blood lead levels below the thresholds at which health effects are produced.

Such an analysis would be fairly complicated, and is beyond the scope of this feasibility study, but could be performed without too great an increase in time or resources. The calculation of background levels of lead in air could be accomplished by the use of monitoring data from the National Air

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[3] 43 Federal Register 52952-53014, November 14, 1978.

[4] Charles River Associates, Economic Impact of the Proposed EPA National Ambient Air Quality Standard for Lead, Cambridge, MA, March 1978.

Surveillance Network; data on the production and use of lead additives in gasoline is easily available, and future production is governed by statutory timetables.

2. Timing of Compliance

One important issue in estimating the effectiveness of the ambient lead standard is the actual timing of exposure and emission reductions. Estimating the extent of exposure or risk reduction over time would require fairly detailed knowledge about the rate at which State Implementation Plans have been or will be revised to deal with lead sources, and the timetables for compliance that are developed. Again, this information is probably available, and could be used to produce a fairly accurate picture of the rate at which emissions are reduced throughout the U.S. Reasonably detailed information about lead emission levels for various industries is also available,[5] and could be used to develop estimates of aggregate emission reductions, which in themselves could be useful measures of effectiveness.

3. Estimates of Exposure Reduction

In order to estimate the effectiveness of the reduction in stationary source lead emissions in reducing adverse health effects, estimates must be developed of the actual changes in air lead levels that would occur around the stationary sources and the number of people that would be effected. The major problem in conducting such an analysis would be the necessity of dealing with each individual point source, or in devising approximations that would enable one to treat similar sources (grey iron foundries, for example) as a class, rather than as individual facilities. The task might be time-consuming, and the results not very precise, but the analysis is certainly feasible. Models for predicting air lead levels around certain classes of stationary source are well-developed,[6] and population densities around individual point sources are generally estimable from Census data. Analysis of this type has been carried out for other pollutants (benzene for example).[7]

For puposes of comparison of this regulation with others (such as the Safe Drinking Water Lead Standard), it is possible to estimate the actual changes in the amounts of lead that are observed in the presence or absence of regulation. The amount of air commonly breathed in 24 hours by adults and children is well known, and the proportion of airborne lead which is absorbed has been extensively studied.[8] It is thus a fairly straightforward exercise to relate changes in airborne lead levels affecting known populations to changes in the total amount of lead absorbed per person by that population. For example, since adults generally breath 15-20 M^3 of air per day, and a number of studies indicate that airborne lead particulates are absorbed with an efficiency of about 35 percent, it is simple to calculate that for each 1 ug/M^3 reduction in airborne lead, an individual will experience a reduction of (15-20) x (.35) x (1) = 5.25 - 7 ug per day in the amount of lead which is absorbed. This calculation can be applied to all of the changes in air lead levels caused by the air lead standard to develop an estimate of the changes in the total amount of lead absorption per person produced by the standard. Given the difficulties involved in calculating changes in the exact number of adverse health effects, which will be described below, this is not a bad measure of effectiveness. It is better than simply tabulating changes in ambient exposure, because the amount of absorbed lead is more clearly related to the production of health effects than exposure. As a crude approximation, an additional increment of absorbed lead, whether from airborne lead particulates or from drinking water, will have a roughly similar effect on the exposed population, in terms of changes in blood lead distribution and health effects. Thus, this measure, total absorbed lead, facilitates comparison between the effectiveness of the two lead regulations, and other regulations limiting general population exposure to lead, from whatever sources.

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[5] Criteria Document, p. 5-3.

[6] Charles River Associates, Economic Impact of the Proposed EPA National Ambient Air Quality Standard for Lead, Cambridge, MA, March 1978.

[7] Stanford Research Institute, Assessment of Human Exposures to Atmospheric Benzene, Prepared for the Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, June 1978.

4. Changes in Blood Lead Distributions and in the Incidence of Adverse Health Effects

The relationship between airborne lead exposure levels and the distribution of blood lead levels has been widely studied, and the intepretation of these relationships, as well as the relationship between blood lead levels and the incidence of adverse health effects has been a major point of controversy in regulatory proceedings both for the OSHA and EPA lead regulations.

The major efforts have been devoted to estimating the relationship between air lead levels and blood lead levels in population groups exposed to different airborne lead levels in different geographical areas.[9] In summarizing the results of various studies, EPA has taken the position that, for each increment of population air lead exposure of 1 ug/dl, the median population blood lead levels will increase approximately 2 ug/dl.[10] Since the statistical properties of population blood lead variability are known,[11] changes in air lead levels can be related to changes in the proportion of the population whose blood lead levels will be elevated beyond certain threshold levels about which the various toxic effects of lead are known to occur. This again enables one to develop effectiveness measures for regulations which result in changes in population exposure to lead. The results of this analysis can be expressed either as changes in blood lead distribution, or as in changes in the number of persons at risk of experiencing the different toxic effects of lead. A sample analysis of this kind will be performed for the Safe Drinking Water Lead Standard.

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[8] Air Quality Criteria for Lead, Chapter 10.

[9] Air Quality Criteria for Lead, Chapter 12.

[10] Preamble to the National Ambient Air Standard for Lead, 43 Federal Register 40246, October 5, 1978.

Changes in the actual number of cases of adverse health effects of lead exposure brought about by the air lead standard cannot be made since, at this time, there are no precise dose-response relationships relating blood lead levels to the incidence of adverse health effects. Recent studies suggest that this is because the actual relationship between the length and intensity of lead exposure, blood lead levels, and the production of adverse health effects is a function of the complex transport and metabolic mechanism of lead in the body.[12] Even so, it is probable that the EPA approach, with few modifications, is an acceptable approximation of the relationship between air lead and blood lead levels in populations where exposure has been relatively constant over time.

e. The Social Cost of Lead - Induced Disease

Since detailed predictions about the changes in incidence of lead-induced disease cannot be developed at this time, the issue of how to estimate the social cost of this disease does not enter in an important way into this analysis. It may enter, however into the calculations of decision-makers trying to compare the effectiveness of the lead regulations with other environmental standards. This issue will be discussed more thoroughly in section A.5 of this appendix.

A.2.2 Costs of the Air Lead Standard

This section of the analysis deals with some of the projected economic impacts of the ambient air quality standard for lead. Quantitative estimates of compliance costs will be developed for several of the more seriously affected industries, and the associated economic impacts of compliance, such as plant closings, unemployment, and effects on international trade, will be briefly discussed.

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[11] Air Quality Criteria for Lead, Chapter 12.

[12] See for example the OSHA Lead Standard Preamble, 43 Federal Register 52952-53014, November 14, 1978.

a. Data Sources and Issues

The two principal sources of data relating to the economic impacts of the air lead standard are both prospective analyses of the standard, performed during the rulemaking proceedings. The first was a study performed on contract to the Lead Industries Association by Charles River Associates.[13] This study develops detailed quantitative estimates of compliance costs for the various lead-processing industries using data on compliance costs supplied directly by the affected firms. The second study was an economic impact assessment performed by the Office of Air Quality Planning and Standards at EPA[14] which estimated compliance cost for some industries. Its main thrust was the prediction of whether or not the viability of any of the affected industry segments would be seriously compromised by the lead standard. EPA apparently used independent engineering estimates for compliance technology costs.

The use of these documents as the primary source of data for estimating the impacts of the air lead standard raises a number of important analytical issues, the first of which has to do with the general problem of the quality of prospective cost estimates. Since these estimates are in fact predictions about the future, they are inherently uncertain. Estimating the degree of uncertainty surrounding such estimates is extremely difficult, and must be based almost strictly on methodological considerations and value judgments. Ultimately, the accuracy of such predictions depends on the accuracy with which existing models can predict future economic behavior, the availability of data to use in these models, and the hope that extraneous and unpredictable factors do not intervene to change economic conditions.

The first two of these factors can be seen at work when one compares the two prospective analyses for the impact of the lead standard, which differ widely in the magnitude of the consequences that they predict. In each case, different assumptions are made about the technological alternatives available to industries to achieve compliance, the extent of compliance which is achievable, and the decision rules that will be used to guide firms in deciding whether or not to continue production. The two analyses also differ in sources of data that they use to estimate costs. It is thus not surprising that the magnitude of economic impacts estimated by the two studies are so different.

The more important sources of uncertainty in the analyses will be discussed as they arise in the course of the development of cost estimates. Attempts will be made to reconcile estimates where that is possible and to at least explain the reason for the differences when such reconciliation is not possible.

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[13] Charles River Associates, Inc., Economic Impact of the Proposed EPA National Ambient Air Quality Standard for Lead, March 1978.

[14] Office of Air Quality Standards and Planning, U.S. Environmental Protection Agency, Economic Impact Assessment for the National Ambient Air Quality Standard for Lead, June 1978.

The availability of retrospective data could alleviate all of these problems. Such data is not available, however. This is because, despite the fact that the regulation was promulgated in 1978, the states have until late 1982 to revise their implementation plans to deal with stationary lead sources. Little of the economic effects of the standard can be expected to have occurred as yet,[15] and what little data we have (as will be discussed later) suggests that few if any of the dramatic effects on the industry predicted by the CRA report have as yet occurred.

b. Estimating Compliance Costs

1. Compliance Scenarios and Baseline Assumptions

Both attempts to estimate the cost of compliance for the airborne lead regulations made a number of important assumptions about the time frame within which compliance would occur, the extent of control that would be necessary to achieve compliance, the degree of compliance ultimately achievable, and the baseline for comparing the regulated and unregulated situations. Since these assumptions have important implications both for the estimate of costs and benefits, they will now be briefly discussed.

Both analyses assumed that all of the capital investment necessary for compliance would occur in 1978, immediately upon promulgation of the standard, and that compliance to the extent feasible with existing (1978) technology would occur immediately thereafter. These assumptions fail to consider the possibility that compliance will probably be undertaken and achieved only gradually, and that control technology may be developed which could reduce the cost of compliance and increase the degree of compliance ultimately achievable. The overall result would be to spread out the capital costs of compliance over time, to increase the degree of compliance (and hence the total benefits) and to reduce the magnitude of control costs.

Both analyses agree that, for some sectors of the industries affected, compliance with the 1.5 ug/m^3 will not ultimately be achievable. The CRA report projects that compliance cannot be achieved by a large portion of the primary lead smelting industry and almost all of the secondary lead smelting industry. The EPA analysis projects that relatively small portions of primary and secondary lead smelting industries will be unable to comply. This has important implications both for the benefits of the standard, since the level of benefits is a function of the degree of exposure reduction, and for the costs of the standard, since firms' decisions about whether or not to undertake control expenditures will undoubtedly be affected by the expected probability of achieving compliance. The cost estimates which are developed assumed that firms will undertake investment, whether or not compliance is achievable. Thus, these compliance cost estimates may be overstated.

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[15] Despite the fact that some firms might hope to obtain competitive advantage by "regulatory anticipation". On the whole, it is in the firms' financial interest to postpone capital investments as long as possible (especially with the prevailing interest rates).

Finally, both analyses assume that in the absence of regulation, firms would not voluntarily undertake any independent efforts to reduce exposure. This is despite the fact that the adverse health effects produced from lead exposure are becoming well-known, and at least some smelter control efforts have been undertaken to reduce emissions independent of regulations.[16/] Making a more realistic assumption about control efforts in the absence of regulation would again lower the costs of reducing lead exposure attributable to the air lead standard.

2. Compliance Cost Estimates

The compliance cost estimates by industry sector developed from the CRA and EPA estimates are summarized in Exhibit A.2-1. It can be seen that CRA only estimated costs for the two major lead processing industries, whereas EPA's estimates include all lead emitting industries.

The cost estimates were adopted slightly from those given in the respective reports, and some extrapolations were made to fill gaps in the analysis, possibly at some cost in accuracy. For example, due to lack of data, CRA did not estimate compliance costs for almost 30% of the primary lead industry. In order to develop cost estimates for the entire industry, it was assumed that the average compliance cost per ton of capacity would be the same for that segment of the industry, as for the other plants for which data was available.[17/] Similarly, for the battery and grey iron foundry industries, it was necessary to assign portions of the industry to the various emission classes developed for model cost estimates by EPA. For the secondary lead industry, the compliance costs for the entire industry were extrapolated on a cost-per-ton basis from cost calculations on a model smelter. The number in Exhibit A.2-1 corresponding to those industries are thus quite imprecise and correspond to reasonable mid-range estimates.

The major issue to be dealt with in interpreting these numbers is the great discrepancy between the CRA and EPA estimates. For the two industries where capital cost estimates were developed by both groups, the CRA estimates exceed the EPA estimates by a factor of about 2.5. This is perplexing, since both groups postulated that essentially the same technology (process evacuation to a baghouse) would be used to achieve compliance. CRA however, used industry estimates of costs at specific facilities, while EPA used engineering cost estimates based on a "model plant" for each industry. Under

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[16/] An epidemiological study of the population surrounding the Bunker Hill Smelter in Kellogg, Idaho, prompted the installation of baghouse filters to reduce lead emissions.

[17/] Such an approach was suggested by CRA (p. 13) but not applied.

EXHIBIT A.2-1

COMPLIANCE COST ESTIMATES FOR THE AIR LEAD STANDARD

Cost Estimates (millions of 1979 dollars)

	Charles River Associates			EPA		
Industry	Capital Costs	Operating Costs	Annualized Costs (20 yrs. @ 15%)	Capital Costs	Operating Costs	Annualized Costs (20 yrs. @ 15%)
Primary Copper Smelting	208.1	16.32	50.42	38.3	-	6.11
Secondary Lead	305.0	30.50	79.21	159.9	-	25.6
Battery	-	-	-	1.2	-	0.19
Grey Iron	-	-	-	112.4	-	16.6
Primary Copper Smelting	-	-	-	319.7	-	56.0
Lead Additives	-	-	-	3.2	-	0.51
Total	513.1	46.82	129.6	634.7	-	105.0

ordinary circumstances we would expect the former methodology to yield more reliable estimates. In this case, however, it must be remembered that both estimates were prepared for use in adversary legal proceedings, and that especially for the firms supplying data to CRA, there was little incentive to develop low cost estimates or consider innovative compliance strategies.

Without detailed studies of the available compliance technologies and cost estimation methodologies, it is difficult to decide which estimate is more accurate. If one had to summarize in one number, the estimated compliance cost for the air lead standard, one might take an average of the cost estimates developed by CRA and EPA for the lead processing industries and add it to the cost estimates developed by EPA for the other affected industries.

3. The Meaning of Compliance Costs

The roughly one-half billion dollar compliance costs estimated in the previous section are not necessarily a good measure of the impact of the lead standard on the economy. These costs do not necessarily reflect a decrease in productive resources, or in total demand. Using welfare economic principals and with sufficient data on the market structure for lead and competing industries, it might be feasible to calculate an economic cost or "dead weight loss" due to the regulation. At this time, insufficient data exists to perform this task. Two other approaches can be taken, however. The first, which will be performed in a later section, is to predict the economic consequences (lost productive capacity, jobs, etc.) associated with the compliance costs. The second approach is to use compliance cost itself as a measure of the relative impact of the standard for purposes of comparison with other regulations.

This can be done (and is done in the final column of Exhibit A.2-1) by converting compliance and operating costs into annualized cost streams which reflect the pre-tax cost of capital and a reasonable time period over which capital costs can be amortized. In this case, an interest rate of 15 percent and a time period of 20 years were selected. The 20 year amortization period is the same period chosen by CRA in their analyses.

The use of annualized costs, while not necessarily reflecting the social costs of compliance associated with the standard does reflect the prevailing market time-preference for investment and thus captures the effect of compliance cost on the cash-flow of affected firms. Annualized costs provides a useful measure of economic impact for the purposes of comparison and aggregation between firms and industries. The next step in the analysis, consideration of the implications of these changes on cash-flow for specific firms and industries will be discussed more thoroughly in the following sections.

c. Economic Impacts of the Air Lead Standard

As discussed in the previous section, estimates of compliance costs are only a part of the total economic impact of any standard or regulation. Intimately associated with compliance costs are a number of other economic effects, direct and indirect, which flow from the necessity to comply with the regulation. Most of these effects are more or less directly tied to the magnitude of the costs incurred by the firms in attempting to comply. Complex models are required to relate the magnitude of compliance costs to the various market decisions undertaken as a result of the added costs. The impact measures most commonly associated with compliance costs are changes in the prices of goods, changes in production volume, and the consequential changes in employment levels and in the balance of international trade. Each of these measures can be related, in theory at least, to some measure of social welfare. But they are also important in their own right, as the various consequences of the regulation may fall disproportionately on specific sectors of the economy or geographic regions.

A major feature of the CRA analysis and, to a lesser extent, the EPA analysis of the economic impact of the air lead standard is the emphasis on the significant number of plant closings that would be required by the infeasibility of compliance. For that reason the discussion that follows is organized around the methodologies used to predict plant closure and the economic impact associated with them. Since the models used to perform these analyses are crude, few if any quantitative estimates of the magnitude of these effects will be made.

1. Predictions of Plant Closures

The Charles River Report predicts that all primary smelters, 80% of all secondary lead smelters and several battery plants will close down as the result of the air lead standard. The EPA analysis predicts, however that at most only a small proportion of the primary smelting industry will shut down. EPA also predicts that only the oldest and "dirtiest" secondary lead and primary copper smelters are in danger of being rendered non-viable by the necessity to comply with the standard. Since we do not have the primary data used by both CRA and EPA to estimate the probability of plant closure, and since the exact structure of the model used by EPA to perform these predictions is not fully described in the documents available to us, it is not likely that we can settle this difference. We can, however, discuss the important general features of the problem and point out how different analytical assumptions can affect the estimated magnitude of economic impacts.

To estimate the likelihood of plant closure, CRA first estimated the likely future "background" concentration of lead around the various facilities due to non-point-source lead emission. If the background lead level in 1982 was greater than 1.5 ug/m^3, the facility was assumed to automatically shut down. In addition, the degree of controls required to reduce lead levels to 1.5 ug/m^3 was calculated taking into account the background lead levels. If the degree of control necessary exceeded the 95% control level achievable with present technology, the plant was again assumed to shut down automatically.

Using these criteria alone, almost all of the plant closures predicted by CRA were made without any consideration of the economic feasibility of compliance.

Clearly this procedure is not the best one to use to accurately predict the actual number of shutdowns due to the standard. Not only does it fail to take into account the possible development of new technology which could allow a greater degree of emission reduction, it also fails to realistically predict the actual compliance process. State compliance plans often are designed in such a way as to reduce adverse economic impacts. It is difficult, for example to realistically believe that a state would require the closure of a battery factory or secondary smelter in a city with a high background lead level due mainly to mobile sources.

Clearly, the assumption that CRA made about the necessity of plant closures are simplistic and tend to overstate the actual likelihood of facilities closure. On the other hand, there is such a large discrepancy in the compliance cost data, that even using the more sophisticated economic models developed by EPA might result in a much larger number of plant closures predicted. There is really no good way to resolve these discrepancies in data without a detailed engineering study or retrospective data.

EPA used a more sophisticated technique to estimate the likelihood of plant closure. Using this compliance cost data, EPA used a discounted cash-flow model to determine if for a number of marginal tax rates and acceptable rates of return, the affected plant would shut down. The exact details of this model were not given in the documents available to us, but as an attempt to model the actual decisions by regulated firms, it is clearly a superior procedure to that used by CRA. Again the model is only as good as its input, and EPA did not have plant-specific data in most cases. EPA also assumed no control technology innovation would occur and that compliance would be instantaneous upon the effective date of the standard.[18] We believe the procedure used by EPA is more defensable than the procedure used by CRA to predict plant closure.

2. Lost Production Capacity

Given that one has predicted a certain number of facilities will close down, the question arises as to how to value the loss of production capacity. Economic theory suggest two alternatives. The first would be to measure the loss of productivity resulting from the shift in capital from the shutdown facilities to the next most productive investment opportunity. In practice this is difficult, since identifying the "next most productive" investment is not always possible. An alternative is to calculate the cost of obtaining the same amount of product (in this case, refined lead) from the least expensive alternative source. Given adequate data about national and international markets, and the costs of constructing new capacity, (whichever is the least expensive alternative) we can, in principal, estimate the actual financial costs of replacing lost capacity. Neither EPA nor CRA attempted to develop quantitative estimates for these kinds of costs, although CRA, in a qualified discussion suggested that extreme price rises could be expected and lead shortages might occur. The magnitude of these effects was likely to have been overstated for the reasons previously discussed.

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[18] It is perhaps difficult for EPA to justify making any other assumption except instant, total compliance.

Aside from the replacement cost of plant closure, there are other associated effects that deserve special mention. Two of these, price changes and loss of employment, are discussed further in later sections. It is important to remember that the economic effects of plant closure are often concentrated regionally. Potentially far-reaching effects can occur when, for example the major industry in a town closes down, reducing tax revenues and local demand for goods and services. Clearly, it is difficult to develop measures for these kinds of costs.

3. The Cost of Unemployment

Each plant closure involves, among other effects, the forced unemployment of the workers at that facility. Aside from the magnitude of the job loss associated with the lead standard (CRA says it will be massive, EPA does not calculate it), the problem arises as to how to develop measures of the impact of unemployment on society. The simplest measure is the number of persons unemployed ("cases" of unemployment) multiplied by the expected length of unemployment, which can usually be estimated. Calculating the expected amount of lost wages, gives one an acceptable measure of the social cost of unemployment. This is because in a perfectly-functioning market, the wage level is equal to the marginal product of that labor. It should be noted, however, that the loss of labor productivity associated with a plant closure could be, to a certain extent, captured in the cost of lost productive capacity described above, so that adding them together in an analysis would be double-counting.

Again, in addition to the aggregate cost of unemployment to society, there are also often distributional aspects to regulation-induced unemployment which could enter in an important way into a decision about how to regulate. For example, in the situation outlined above, where the principal employer in a town or region shuts down, the decrease in demand for goods or services may cause other workers to be laid-off. In addition, depressed economic conditions in a region, coupled with a high unemployment rate could act to depress wages for those still employed.

Finally it is likely that unemployment has social consequences not commonly captured by social accounting mechanisms about which we may be concerned. Such externalities could include lost self-respect and self reliance on the part of the unemployment, reduced standards of living, decreases in family stability, and increases in crime and disease. Clearly, some notice needs to be taken of these effects, yet they are extremely hard to quantify and include in an analysis in any consistant manner.

4. Price Changes Due to Regulation

The classical economic analysis of the effects of regulation on social welfare treats the regulation as if were a tax which raises and possibly changes the shape of the industry supply curve for the regulated good. The changes in producers' and consumers' surplus and a "dead-weight-loss" due to regulation can then be calculated. Such analyses are often more difficult than they appear, because of the lack of data as to the slope of the supply and demand functions for the particular good in question. Such analyses usually comprise an important part of the analysis of regulatory impacts. Few data exist that would enable us to apply such an analysis in this case, and neither CRA nor EPA attempted to conduct such an analysis for the air lead standard. For that reason, that kind of analysis will not be discussed further here.

d. Conclusion

The major problem encountered in developing estimates of the economic impacts of compliance with the lead standard was resolving the great differences in compliance costs estimated by the two prospective studies of the standard. With little retrospective data to guide us, it was difficult to resolve many of the conflicts between the two estimates of compliance costs, which differed by a factor of 2.5 for the two industries studied by both groups. It would appear that the total capital investment required for compliance will be somewhere between 634 million and 950 million dollars. Both of these estimates are highly uncertain. It is possible that they both over estimated compliance costs since they each preclude the possibility of control technology development. The total annualized costs to industry, including operating cost estimates where they are available range between 105 million and 202.9 million dollars.

We did not develop quantitative estimates for any other of the economic impacts of the standard, although some of the theoretical and practical difficulties of estimating the magnitude of the impacts were discussed.

A.3 PRIMARY DRINKING WATER STANDARD FOR LEAD

A.3.1 Effectiveness of the Drinking Water Standard

a. Existing Risk Assessments

There was no quantitative assessment of risk reduction undertaken by EPA prior to the promulgation of the primary drinking water standard for lead in 1975. Instead, the pollutant control levels for inorganic pollutants were apparently set on the basis of consensus standards set by the Public Health Service in 1962.[1] In 1977, a major study of the health effects of trace contaminants in drinking water by the National Research Council,[2] which had been mandated by the Safe Drinking Water Act, was published. No quantitative estimates of risk were undertaken, but the health effects of lead exposure were reviewed, the population at risk was estimated, and a suggestion was made that the 50 ug/l standard "may not ... provide an ample margin of safety."[3] The standard was not revised, however, when the Primary Drinking Water Standard were amended in 1979, and the maximum allowable level remains at 50 ug/l.

b. Feasibility of Risk Reduction Assessment and Effectiveness Estimation

Many of the same factors which affected the feasibility of the risk reduction assessment for the National Ambient Air Quality Standard for Lead also affect the feasibility of conducting such an analysis for the Safe Drinking Water lead standard. This is especially so in the later stage of the analysis, where the same problems are encountered in determining the relationship between lead exposure levels and changes in blood lead level, and between blood lead levels and the incidence of lead-induced toxic effects. Also, as will be seen, there is the question of how to take into account the effects of other regulations which also affect lead exposure in the general population, and provide a background against which the drinking water standard must be evaluated.

Different kinds of problems are encountered in the early stages of the analysis, however. Calculating the population at risk is rather difficult because little data exists as to extent to which lead pipes (almost the exclusive cause of excess lead levels in drinking water) are in use. In addition, risk reduction is difficult to calculate because little data is available as to the extent to which lead pipes are being replaced as a result of the standard, and less data is available as to the rate at which such pipe would have been replaced in the absence of the standard.

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[1] United States Public Health Service, 1962 USPHS Drinking Water Standards, PHS Publication #956.

[2] National Research Council, National Academy of Sciences, Drinking Water and Health, Washington, D.C., 1977.

[3] Ibid, p. 261.

On the whole however, we believe that a risk reduction estimate, as described in the discussion of the ambient air lead standard, whose outcome measures are changes in the amount of lead absorbed by the population at risk, and in the number of persons at risk of lead-induced toxic effects, is feasible. That is, although the sample analysis which follows makes a number of heroic assumptions, data should be available, given further resources, to considerably narrow the degree of uncertainty in the final risk reduction estimate.

c. Analytical Issues: Sample Risk Reduction Estimate

This discussion is organized around the performance of a risk-reduction assessment as described in section A.2.1. The analysis is used to illustrate important analytical issues.

1. Joint Benefits with the EPA Standards Regulating Lead in Air and in Gasoline

The calculation of the effectiveness of the drinking water regulation in reducing the total amount of lead absorbed by the population at risk is not affected to any great extent by changes in air lead levels brought about by the other EPA lead standards, although it is possible that slight changes in the lead content of surface water might marginally affect the lead intake of the exposed population. The other two lead standards could, however, affect the results of the blood lead analysis, since, over time, reductions in air lead could produce reductions in background blood lead levels. For this analysis, however, we will assume that between the year 1980 and 2000 the blood lead levels of the urban and rural populations do not change, but remain constant (in the absence of lead exposure from drinking water) at 10 ug/dl and 18 ug/dl, respectively, for both adults and children. These values are typical of the average urban and rural blood lead values in the absence of abnormally high exposures, as summarized in the Air Quality Criteria Document for Lead.

2. Population at Risk and Timing of Compliance

As stated above, there is very little data available in the literature to indicate the extent to which high lead levels in drinking water are a problem. The one quantitative study we found, which was based on a 1969 survey of 99 water systems conducted by the Public Health Service, estimated that about 2% of the survey population (about 18 million people) were exposed to levels above 0.05 mg/l in their drinking water.[4]

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[4] McCabe, L.J., et al.," Survey of Community Water Supply Systems" Journal of the American Water Works Association 62, 670-687, 1980.

Extrapolated to the entire population of the United States, this implies that in 1970, about 4 million people were exposed to greater than 50 ug/l lead in their drinking water.

The question then arises as to the rate at which lead pipes in water distribution systems are being replaced, and at what rate they would have been replaced in the absence of the drinking water standard. Our limited discussions with water distribution engineers and managers suggest that, while it is generally known that lead distribution pipes are the main cause of high lead in drinking water, few if any municipalities have engaged in systematic efforts to replace lead pipes for that reason. The promulgation of the drinking water standard has not resulted in a dramatic increase in the rate at which pipes are being replaced. Thus, in order to both establish a baseline of what would have occured in the absence of regulations, and what has occurred since, the first of our "heroic" assumptions had to be made; namely, that in the absence of regulation, lead pipes would have been replaced at a rate of 1 percent per year, and that because of the regulation, they would instead be replaced at a rate of 5 percent per year. An extensive data gathering effort aimed at municipalities known to have the problem could document the extent to which pipes actually have been replaced, and aid in making far more informed predictions about the rate of lead pipe replacement in the future.

Using our assumptions about baselines and the compliance rate, it is possible to make rough estimates of the number of people who will be exposed to excessive levels of lead due to lead water pipes over the next two decades. These estimates, based on population trends over the last 10-15 years taken from the Statistical Abstract of the United States, 1979 are summarized in Exhibit A.3-1. Estimates are given for both adults and children under age five because the latter group is known to be especially sensitive to lead exposure.

3. Reductions of Exposure and Absorbed Lead

Having estimated the change in the number of persons exposed to excessive lead levels in drinking water, two more pieces of information are necessary if the amount of lead absorbed by the entire exposed population is to be estimated. The first piece of information needed is the actual level of lead found in areas served by lead pipes. A more precise estimate could probably be obtained by going back to the original Public Health Service Survey, but for our purposes we will use a study performed on several areas in Cambridge and Somerville, Massachusetts[5] where lead pipe use is very common. This study found that the average lead level in drinking water as 93 ug/l, roughly twice the EPA standard. We will assume, for the purposes of this analysis, that this is a typical value for the lead content in drinking water in lead pipe areas.

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[5] Karalekas, P.C., et al, "Lead and Other Trace Metals in Drinking Water in the Boston Metropolitan Area", Journal of the New England Water Works Association, 90, 150-172, 1976.

EXHIBIT A.3-1

ESTIMATES OF TOTAL U.S. POPULATION EXPOSED TO EXCESSIVE LEVELS OF LEAD IN DRINKING WATER, 1980-2000

	NO DRINKING WATER REGULATION (1)		WITH DRINKING WATER REGULATION (1)	
	ADULTS	CHILDREN (3)	ADULTS	CHILDREN (3)
1980	4,110,000	272,000	2,950,000	196,000
1985	4,060,000	245,000	2,380,000	151,000
1990	4,010,000	218,000	2,010,000	110,000
1995	3,970,000	197,000	1,630,000	81,000
2000	3,930,000	178,000	1,300,000	59,000

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(1) Assuming lead pipes replaced at 1 percent per year.

(2) Assuming lead pipes replaced at 5 percent per year.

(3) Five years old and under.

The next piece of information we need is how much lead is actually absorbed by persons exposed to high-lead water. There is much data in the literature covering this point. A number of studies have established a consensus that the average drinking water intake of adults and children under 5 are about 2 liters and 0.5-1.0 liters, respectively.[6] In addition, studies have indicated that lead from drinking water is absorbed into the body with an efficiency of about 50 percent.[7] Using these figures, it is straightforward to estimate the yearly lead intake from water as follows:

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[6] _Drinking Water and Health_, p. 11.

[7] _Air Quality Criteria for Lead_, p. 12-33.

Adults

(93 ug/l) x (2 l/day) x (0.5) x (365 days/year) = 33945 ug/year

Children

(93 ug/l) x (0.5 l/day) x (0.5-1.0) x (365 days/year) = 8486.25- ug/year
16972.5

Using these estimates, and the previously developed projection of the number of people at risk, it is straightforward to estimate the change in total lead absorption brought about by the drinking water standard. The results are summarized in Exhibits A.3-2 and A.3-3, where it can be seen that, using the analytical assumptions described above, the regulation appears to reduce rather dramatically total lead absorption, both for adults and children. Summing all the exposure reduction for the years 1980-2000, it is predicted that the drinking water standard will reduce total amount of lead absorbed by adults from 2860.7 kilograms to 1440.3 kilograms. The amount of lead absorbed by children will be reduced from 39.4 to 20.5 kilograms. Using this measure of effectiveness of the drinking water standard has two advantages over measures of reduced exposure. First, it is a measure of the change in lead that is actually active in producing toxic effects, and second, similar measures can be developed for other lead regulations so that effectiveness can be compared across regulations. In the next section we demonstrate how this analysis can be carried one step further to relate changes in absorbed lead to changes in blood lead levels.

4. Changes in Blood Lead Levels

Blood lead levels are, in and of themselves, a crude indicator of the degree of lead exposure experienced by individuals or populations. In addition, lead levels can be related to the production of toxic effects of lead, in some instances. Thus, the changes in blood lead levels brought about by regulations constitute a crude measure of risk reduction.

In order to estimate changes in blood lead levels brought about by changes in exposure, it is necessary to make a series of assumptions about blood lead levels and distribution in the absence of exposure, and the relationship between lead exposure, absorption, and change in blood lead levels. For the purposes of illustration, the analysis is shown only for children. We adopt a set of assumptions made by EPA in the preamble of the Ambient Air Quality Standard for Lead[8] or that are consistent with the literature as summarized in the Criteria Document. The basic assumptions are:

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[8] 43 Federal Register 46246, October 5, 1978.

EXHIBIT A.3-2

ESTIMATED TOTAL POPULATION LEAD ABSORPTION (KILOGRAMS) DUE TO LEAD PIPES, 1980-2000 (ADULTS)

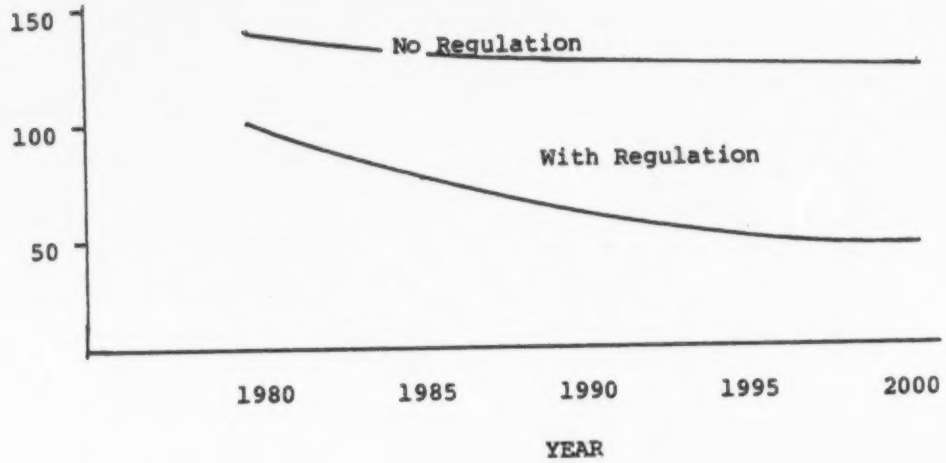
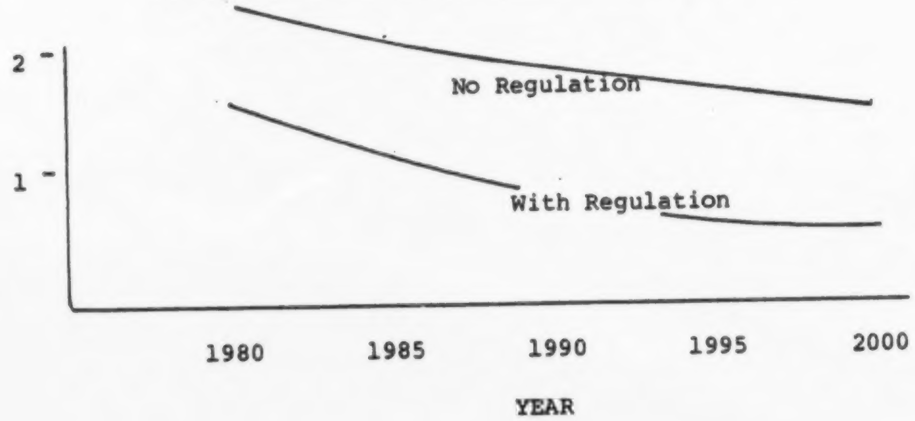


EXHIBIT A.3-3

ESTIMATED TOTAL POPULATION LEAD ABSORPTION (KILOGRAMS) DUE TO LEAD PIPES, 1980-2000 (CHILDREN)



1. For children less than 5 years old, the median blood lead levels in the absence of lead from drinking water are 10 ug/dl in rural areas and 18 ug/dl in urban areas.

2. Between 1980 and 2000, roughly 75 percent of all children will live in urban area, 25 percent in rural areas.

3. Blood lead levels are distributed with a geometric standard deviation of 1.3 ug/dl about the median values.

4. EPA states that each 1 ug/M^3 of airborne lead increases the median blood/lead level in the exposed population by 2 ug/dl. Each 1 ug/M^3 exposure produces an increase in the amount of absorbed lead of about 7 ug per day (assuming 20 M^3 air breathed and 35 percent absorption efficiency). Thus, assuming that children drink 1.0 liter of water per day, and absorb about 50 percent of the lead, the amount of lead absorbed from drinking water with an average lead level of 93 ug/l is approximately enough to raise the median blood lead level about 13 ug/dl, which corresponds to a median blood level of 23 ug/dl in rural populations and 31 ug/dl in urban populations.

Given that the increase in the median blood lead levels can be calculated and that the number of children at risk has been estimated, the number of children whose blood lead is above harmful levels can be calculated, using the statistical properties of the blood lead distribution (assumption 3). The results of these calculations are summarized in Exhibits A.3-4, A.3-5, and A.3-6. Again, it can be seen that the drinking water standard will produce substantial decreases in the number of children with blood lead levels above 40, 50, 60 and 80 ug/dl.

The results of this analysis, resting as they do on a number of assumptions made in the face of both a great shortage of data, and a lack of precise consensus as to the physiologic behavior of lead-exposed populations, must be considered to be most imprecise. Further effort could refine many of these analytic assumptions, and replace several of them with reasonably hard data, thus reducing the degree of uncertainty. In particular, data could probably be gathered as to the extent of exposure reduction, and more sophisticated physiological models could be used to relate lead exposure to changes in lead levels. This example does, however, illustrate the feasibility of conducting analysis which relate regulatory programs to changes in measurable physiological parameters which are themselves clearly related to health effects. Similar analysis could also be conducted for the ambient air lead standard and for the standard controlling lead levels in gasoline.

5. Changes in Incidence of Lead-Induced Disease

The changes in blood lead levels described in the previous section can be related in a general way to the frequency of incidence of lead-induced disease. According to a general consensus of investigators, each of the blood lead levels, 40, 50, 60 and 80 ug/dl are threshold levels, below

EXHIBIT A.3-4

NUMBER OF CHILDREN WITH BLOOD LEAD LEVELS GREATER THAN 40 ug/dl

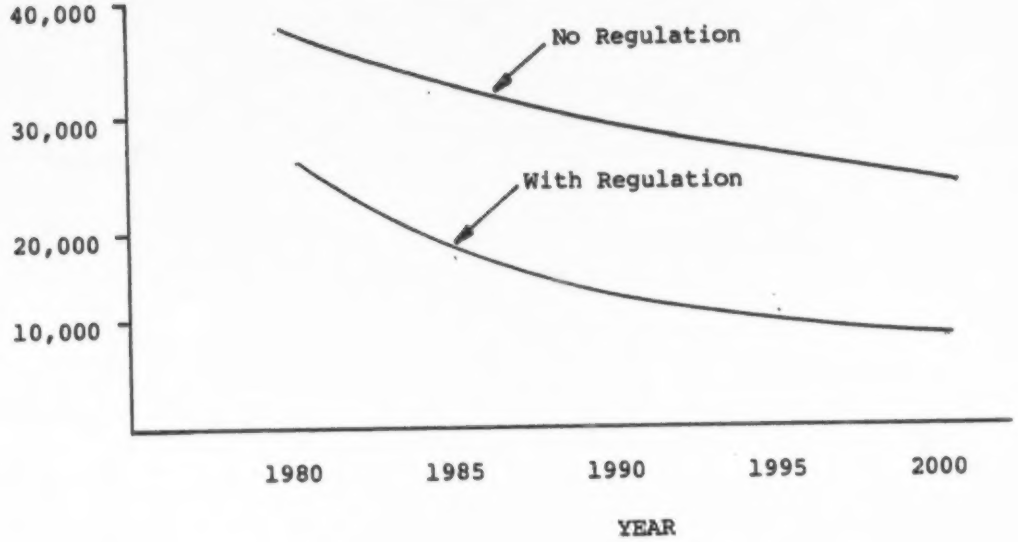


EXHIBIT A.3-5

NUMBER OF CHILDREN WITH BLOOD LEAD LEVELS ABOVE 50 ug/dl AND 60 ug/dl

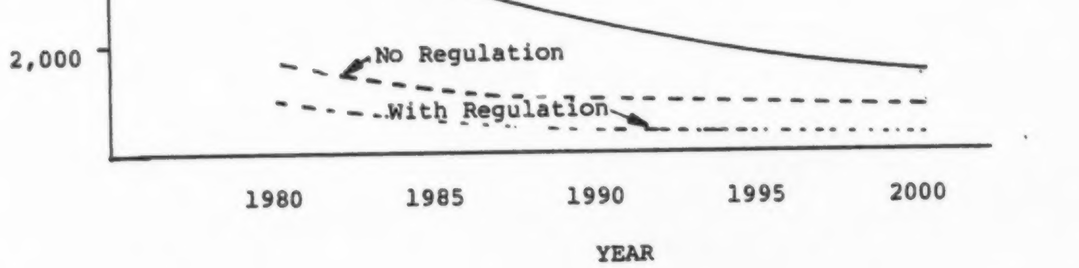
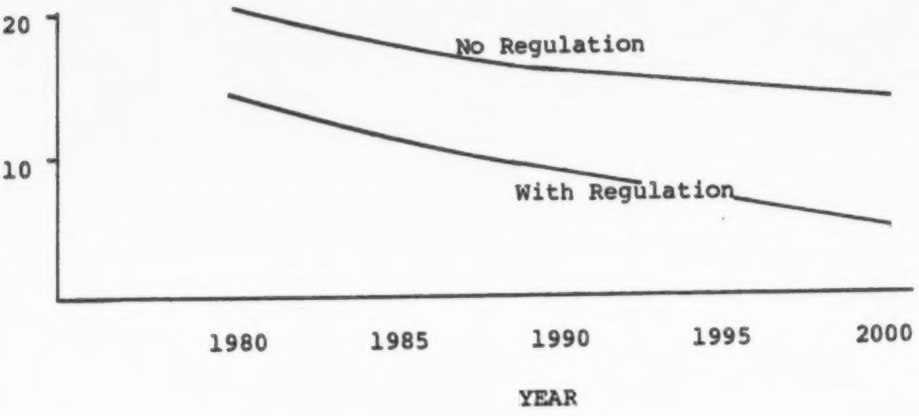


EXHIBIT A.3-6

NUMBER OF CHILDREN WITH BLOOD LEAD GREATER THAN 80 ug/dl



which certain toxic effects of lead have not been observed to occur.[9] For example, lead-induced anemia has not been observed to occur at blood lead levels below 40 ug/dl. Detriments in cognitive functions and peripheral neuropathy (two distinct syndromes) do not seem to occur below 50 or 60 ug/dl. Lead encephalopathy, an acute and often fatal disorder of the central nervous system, which almost always results in permenant mental impairment in survivors, is not thought to occur below 80 ug/dl in children.

Thus, the reductions in the number of children with blood lead levels above the various levels brought about by the regulations, can be thought of as reductions in the number of children at risk of suffering these various disorders. At present, it is not possible to make estimates of the number of children that will actually suffer these disorders, primarily because the incidence of these diseases depends on many other variables, including individual variations in susceptibility, socioeconomic factors, and length of exposure. In particular, the incidence of one important disease syndrome produced by lead exposure, progressive kidney failure, seems to depend in an important way on both the intensity and length of exposure.[10] In addition, there are other toxic effects of lead, on the cardiovascular, endocrine and immunologic system about which not enough is known to allow any kind of relationship to be developed between blood lead and incidence. Similarly, whether there is a relationship between blood lead levels and the induction of cancer is not known at this time. If this analysis were undertaken on an adult population, the reproductive status of the exposed population would have to be taken into account, owing to the known effects of lead on the reproductive system.

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[9] Air Quality Criteria for Lead, p. 13-8.

[10] Air Quality Criteria for Lead, p. 11-45.

It does seem, however, that the change in blood lead level distribution brought about by the drinking water standard (or other lead standards), when understood in the context of threshold levels for the production of harm, affords an excellent measure of the effectiveness of the regulation. If for example, one wishes to define an effectiveness goal of preventing all adverse health effects to the population, the reduction of the number of persons with blood lead levels above 40 ug/dl is not an unreasonable choice, since few clinical symptoms of lead poisoning occur below this level.[11] In setting the ambient air quality standard for lead, EPA used a variation of this rationale to justify the reduction of the ambient level.

6. Social Costs of Lead-Induced Disease

As stated previously, since it is not possible to develop quantitative measures of the change in the incidence of lead-induced disease brought about by regulation, developing monetary estimates of the health impacts of the lead regulation is also not feasible. It should be stated, however, that many of the effects of lead exposure are in fact subtle, and not readily amendable to quantitative treatment.

A.3.2 Costs of the Drinking Water Standard

Lead in drinking water does not come from natural sources but from either lead in pipes or the solder used to connect them. A PHS survey of the levels of lead in drinking water revealed that higher lead levels appear in older water systems in several U.S. cities. Although the adverse health effects of lead in drinking water are well documented, there has been no attempt that we could find to estimate the cost of compliance with the lead standard in drinking water.

We contacted water distribution engineers working for several municipal water systems. The engineers were asked if lead levels in the water exceeded the standard, and if so, what would be the cost of compliance with the standard. While they could discuss the controls necessary for compliance, they could not estimate the costs. The corrective measures the engineers cited were replacement of old pipes serving the populations from that water system, and adjusting the pH of the water which decreases the amount of lead leached from the pipes. All the engineers commented that the most effective way to reduce lead in water was to remove and replace the lead pipes and joints. For purposes of this analysis, we assume that replacement of the lead pipes and joints are required to meet the standard.

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[11] There is still some controversy as to the clinical significance of blood lead level elevations below 40 ug/dl.

In order to estimate the cost of compliance of the lead standard in drinking water, two types of data are necessary. The first is the cost of replacing lead pipes and lead joints in a city with a high lead level in its drinking water. The second is to identify the cities where the lead level is high in drinking water. This data could be obtained, but is beyond the scope of this feasibility analysis. The Bureau of Water Hygiene of the Public Health Service published, "The Community Water Supply Survey" in 1970. The survey would serve as a guide in identifying those cities with high lead levels in drinking water. An estimation of lead pipe and joint replacement costs could be made by interviewing those who are associated with water management and plumbing construction for municipal use. The estimate would be based upon the magnitude of the national problem (number and size of water systems with lead levels) and the costs associated with replacement.

A.4 THE FEDERAL WATER POLLUTION CONTROL ACT SECTION 311 LEAD STANDARD

A.4.1 Introduction

The lead standard authorized by Section 311 of the Federal Water Pollution Control Act and codified in 40 CFR 117 establishes reportable quantities for lead, describes requirements for notification in the event of a discharge; and prescribes penalty provisions. The lead regulation is intended to encourage a high standard of care in handling this substance by prohibiting its discharge above the reportable quantity. Exhibit A.4-1 lists the lead compounds covered by these regulations and the reportable quantity for each.

EXHIBIT A.4-1

LEAD COMPOUNDS SUBJECT TO REGULATION
UNDER SECTION 311 OF THE FEDERAL WATER POLLUTION CONTROL ACT

Common Name	Synonyms	Reportable Quantity in pounds
Lead acetate	Sugar of lead	5,000
Lead arsenate		5,000
Lead chloride		5,000
Lead fluoborate	Lead fluoroborate	5,000
Lead fluoride	Lead difluoride, plumbous fluoride	1,000
Lead iodide		5,000
Lead nitrate		5,000
Lead stearate	Stearic acid lead salt	5,000
Lead sulfate		5,000
Lead sulfide	Galena	5,000
Lead thiocyanate	Lead sulfocyanate	5,000
Tetraethyl Lead		100

The purpose of this case study is to analyze the feasibility of conducting a cost-effectiveness analysis for this lead standard. Data sources, data availability, and analytical issues pertaining to the effectiveness side of the analysis will be discussed first. A similar discussion regarding costs will follow.

A.4.2 Benefits

No comprehensive analysis of the benefits of this lead standard exists. Therefore, in order to include this standard (and others promulgated under Section 311) in a cost-effectiveness analysis, an analysis of benefits would need to be performed.

To measure the benefits of the lead standard, we need to know the frequency and volume of spills over the reportable quantity which were committed before and after the regulation was implemented and how much of any net decrease can be attributed to the 311 regulation. We also need to examine whether notification procedures minimized harmful exposures to lead because clean-up was initiated earlier (since spills were reported earlier). By reporting a discharge immediately to the Coast Guard, appropriate action can be taken, in theory, as soon as possible to mitigate damage to human health. Similarly, clean-up costs for reported spills may be reduced by these regulations, since clean-up may be initiated earlier and consequently, may not require as extensive or expensive procedures. Furthermore, to measure benefits, we need to evaluate how much harmful exposures were reduced by the lead discharge standard.

The lead standard requires the person in charge of a source of discharge to notify by telephone the National Response Center (NRC), U.S. Coast Guard, immediately upon discovering a discharge. An NRC duty officer receives the telephone call and collects specific information about the time and place of the spill, the quantity (if known) of lead discharged, the name of the reporter, etc. These incident reports have been officially collected since September 29, 1979 (and since November 20, 1980 for discharges from common carriers). The reports are kept on file by the National Response Center. By looking through the approximately 10,000 reports per year on file, we could assess the frequency and volume of lead spills over the reportable quantity which were committed after the lead standard was implemented. The reports are organized sequentially, so it would be necessary to review all the reports to find the ones pertinent to the lead standard.[1]

Once we have collected the post-regulation data, we would need to collect the same type of data for pre-regulation discharges. The National Response Center has incident reports on file for discharges prior to September 29, 1979 only for those discharges "voluntarily reported or fortuitously discovered."[2] Consequently, the files contain reports primarily for massive spills and spills voluntarily reported by individuals because of their conspicuous nature (i.e., strong odor or blatent color in water). Hence, the incident reporting data for discharges prior to September 1979 is incomplete and does not cover all discharges.

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[1] J. Amson of the Marine Activities Branch, EPA performed this task in 1980 and estimated that the time necessary to complete the task was 30 hours per year of reports. Discussions with J. Amson, February 17, 1981.

[2] Discussions with J. Amson, Marine Activity Branch, EPA, February 17, 1981.

To compensate for the incomplete manner of spill reporting, one study assumed that pre-regulation reported spills accounted for only 10 percent of the actual number of discharges.[3] This assumption was justified by the experience of oil spill reporting which increased by at least ten-times over the voluntary oil spill reporting after mandatory reporting requirements were established in 1970 under Section 11(B)(4) of the Water Quality Improvement Act. Another study assumed that only 20 to 25 percent of actual incidents were reported prior to the mandatory requirements.[4] This inconsistency points out the inexact nature of such extrapolations.

Another way to collect information on the historical trends of lead spills is to use the U.S. Coast Guard's Pollution Incident Reporting System (PIRS) data file. This file contains discharge data for every spill reported since 1971 and response and penalty data for every spill reported since 1973. Exhibit A.4-2 lists the type of data on file.

We have also reviewed a PIRS listing of all spills of hazardous materials from vessels for which reports have been received from 1973 through 1977. Out of the approximately 480 reports on file, only one was for a lead spill. Because of the infrequency of reported lead spills, the lead standard may not be representative of discharge regulations promulgated under Section 311. However, it is otherwise representative of the difficulties in developing benefits estimates for Section 311 standards.

The PIRS data listed on the lead spill is presented in Exhibit A.4-3. The Coast Guard cautions that the PIRS listing should not be considered indicative of the spill rate of hazardous substances in the U.S. because the listing includes only spills which have been voluntarily reported or fortuitously discovered by the Coast Guard.[5] It is important to note that the two gallons (or 16 pounds) of lead spilled is much less than the reportable quantities for lead listed in Exhibit A.4-1. As a result, this lead spill, most likely, would not have been reported under the requirements of the lead standard.

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[3] Jonathan E. Amson, "An Analysis of the Economic Impact of Hazardous Substances Regulations," Control of Hazardous Materials Spills -- Proceedings of 1976 National Conference on Control of Hazardous Material Spills, New Orleans, 1976, p. 49.

[4] George J. Moein, "Magnitude of the Chemical Spill Problem: A Regional Overview", Control of Hazardous Material Spills -- Proceedings of 1978 National Conference on Control of Hazardous Material Spills, Miami, 1978.

[5] Discussions with Lieutenant Paul Fulton, U.S. Coast Guard, February 27, 1981.

EXHIBIT A.4-2

TYPES OF DATA AVAILABLE FROM PIRS FILES

I. Discharge
- time
- location
- state
- waterbody
- source
- source identifier
- cause
- operation
- material
- quantity
- affected resources
- weather
- notifier
- anticipated response
- Coast Guard unit's operating facility number

II. Response
- removal party
- equipment used
- personnel
- duration of response
- amount recovered
- cost of cleanup

III. Penalty Action
- penalty action initiated
- initiating agency
- authority
- action taken against party
- action date
- referral to U.S. attorney
- referral to Commandant/other agency
- action by U.S. attorney
- penalty, fine or settlement assessed
- imprisonment
- suspension, revocation, or probation
- hearing or trial
- first appeal
- second appeal
- civil action appealed to U.S. court
- penalty, fine, or settlement collected
- case closed

EXHIBIT A.4-3

SAMPLE INFORMATION FROM A PIRS DATA FILE: LEAD SPILLS

Substance:	Lead Compounds
Substance Code:	2153
Date:	March 1, 1977
Location:	River area, channel or other restricted navigable waterway in California
Source:	Tankship, 35,000 to 49,999 gross tons.
Quantity:	2 Gallons
Cause:	Personal Error--Unintentional Discharge
Type of Operation:	Vessel-Related Operation--Receiving Dry Cargo

To determine whether and by how much the notification procedures reduced harmful exposures to lead by initiating clean-up actions earlier, we would need pre- and post-regulation data on actual human exposures to discharges of lead. This may be attainable for post-regulation discharges from NRC files using the time-consuming procedure described above or from PIRS files. Pre-regulation data, however, will be limited for the reasons specified above.

Another possible benefit of the discharge regulation is the reduced monetary loss due to disruption of the production processes if clean-up is initiated earlier. Not all spills cause a disruption of the production process, but in the events of toxic fumes, fire hazards, or other life-endangering events, plant operational shut-down is often required during cleanup efforts. Loss of production time, clearly, is a cost of hazardous substance spills, but if clean-up is initiated earlier as a result of discharge reporting, then the loss of production time may be minimized. In this way, regulations under Section 311 may reduce the monetary loss due to the disruption of production processes.

Additional benefits of earlier clean-up actions include:

- decreased costs of plant equipment and property damaged or destroyed,
- decreased compensation and/or costs of injury to plant workers,
- decreased compensation for damage to non-plant property (e.g., airborne vapor or dust carried downwind of a spill incident, resulting in destruction to buildings, trees, shrubbery, etc.).

A.4.3 Costs

The types of direct costs resulting from the lead discharge standard include:

- fines and penalties paid for illegal discharges,
- notification costs,
- clean-up and damage mitigation costs,[6] and
- precautionary costs (spill prevention costs).

Available notification, clean-up, and spill prevention data is limited to actual cost estimates made by industry. Records of fine schedules and penalty histories for past discharges are available from PIRS data files, but the data for discharges prior to September 1979 are limited.

Two publications analyzing the costs of hazardous materials' discharges were reviewed:

(1) "An Analysis of the Economic Impact of Hazardous Substance Regulations", and

(2) "An Analysis of the Economic Impact of Spill Prevention Costs on the Chemical Industry."

Both were written for the Hazardous Materials Spill Conference Proceedings by Jonathan Amson in 1976 and 1978, respectively.

The first study summarizes 1973 to 1977 cost data for three categories of hazardous materials (e.g., soluble chemicals, insoluble floating "oil like" chemicals and insoluble sinking chemicals). The spill prevention cost study describes methods of estimating industry investments into spill prevention as a result of discharge penalties. These studies would be useful, but not comprehensive, sources of regulation cost data.

One glaring problem for comparing pre- and post-regulation costs is that as incident reporting requirements have been issued, clean-up technology has developed, and consequently, costs of the technology have increased. Comparisons would have to be adjusted for this factor.

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[6] As discussed earlier, reduced clean-up costs may be a benefit of the regulations if, as a result of the mandatory reporting requirements, clean-up is initiated earlier. Alternatively, the lead discharge standard could result in increased clean-up costs for spills which would not have been reported voluntarily prior to the regulations, but under the mandatory reporting requirements, must be reported and ultimately cleaned-up.

A.4.4 SUMMARY

The lead standard is representative of the difficulties of estimating benefits and costs of section 311 standards. Data for the lead discharge standard and for other discharge standards are limited especially for pre-regulation spills. Furthermore, determining the effectiveness of discharge standards in terms of improved human health may be a near insurmountable problem. It may be possible to determine the change in frequency and volume of spills as a result of the regulation. However, the magnitude and severity of human exposures to discharged pollutants depends on the characteristics of the receiving waters. Examining all possible such receiving waters in this regard would be a very large project.

A.5 COST-EFFECTIVENESS ANALYSIS OF LEAD REGULATIONS

A.5.1 Cost and Effectiveness Measures for the Three Lead Regulations

The data needed to develop quantitative measures of costs and effectiveness are available to a different degree for each of the three regulations discussed in the previous sections of this appendix.

Extensive industry participation in rulemaking proceedings and explicit consideration of the economic impact (although not required) lead to the development of fairly detailed industry-specific estimates of costs for compliance with the National Ambient Air Quality Standard for Lead. Less information was available however, concerning the likely costs for the replacement of lead water distribution pipes which would be required to achieve compliance with the Primary Drinking Water Standard. Little data has been gathered even to the exact extent to which lead pipes have been replaced as a result of the drinking water standard. Even less data is available to allow the determination of the costs of complying with the section 311 spill control and reporting regulations. What data there is (reported spills and fines assessed) has not been organized, so that estimating these particular costs would be time-consuming. Other compliance costs for the standard (reporting costs and investments in spill prevention equipment) have been estimated for the entire 311 program, but not specifically for lead spills. It would be very difficult to attribute some portion of these costs to lead spills, without knowing the frequency and severity of lead spills, relative to other substances.

Quantitative estimates of the economic impact of the air lead standard were developed in section A.2. Prospective analyses conducted by consultants for industry and for EPA allowed detailed estimates of industry compliance costs to be developed, and allowed less precise estimates of the magnitude of other economic impacts to be made as well, including effects on production, employment, prices and international trade.

It is likely that with more research, reasonably accurate quantitative estimates of the costs of replacing lead pipes could also be made. In this case, however, the secondary economic impacts could be quite different, however, since for example, the cost would probably fall on the municipal and federal governments and effects on employment might be positive rather than negative. The selection of cost measures may not be any more difficult in theory for the section 311 regulations, but data do not permit quantitative measures to be developed at this time.

Measures of the extent of risk reduction were found to be feasible also for both the ambient air quality standard and the drinking water standard. Although no previous quantitative assessments of risk reduction had been performed, data exists which allows the estimation of changes in blood lead distributions in populations at risk brought about by the two standards. In practice, risk reduction estimates were easier to perform (with the help of a few simplifying assumptions) for the drinking water standard, and quantitative estimates were developed for changes in blood lead levels due to the replacement of lead pipes. Performance of a similar analysis for the ambient air quality standard would have required the application of air dispersion modelling to a large number of individual point sources, and estimation of changes in population exposure around each source. While this analysis is quite straightforward, the amount of effort required to perform it is beyond the scope of this feasibility study.

The measures of risk assessment chosen, reduction in total lead absorbed by the population at risk, and changes in blood lead levels in exposed populations, fall short of providing actual measures of the health impact of the lead standards, in terms of reduced incidence of disease. Since, however, there is fairly broad consensus that the various health effects of lead exposure do not occur below certain blood lead levels, the changes in blood lead levels can also be interpreted to be changes in the number of people at risk of, for example, anemia, or potentially fatal lead encephalopathy. Thus, the change in blood lead levels especially lends itself well to being used as an effectiveness measure, assuming one's goal is to prevent occurrence of lead-induced disease. For example, one could set one's goal at totally preventing the occurrence of lead encephalopathy (which does not seem to occur at blood lead levels below 80 ug/dl) or select a goal of allowing no more than a certain proportion of the population to experience blood lead levels greater than 40 ug/dl, the level at which the first overt clinical symptoms (anemia) appear in some persons. A variant of this approach was taken by EPA in justifying the selection of the 1.5 ug/M^3 standard for ambient air levels.

Blood lead levels also lend themselves nicely to marginal effectiveness analysis. It is quite straightforward to determine the changes in blood lead distribution, and correspondingly, in the population at risk of lead-induced disease, for varying degrees of control.

A.5.2 Feasibility of Cost and Effectiveness Comparisons Among the Three Lead Regulations

Two of the three lead regulations under study could be subject to detailed comparison of costs and effectiveness, with relatively little additional effort. The costs of the air lead standard and the safe drinking water standard could be so compared. Some additional work is required before these comparisons can fully include some of the secondary economic effects of compliance, such as effects on employment or international trade. It is to be anticipated, however, that many of the adverse economic impacts associated with the air lead standard (lost employment, negative effects on international trade and so on) would not occur as a result of compliance with the drinking water standard, because such compliance would not involve expenditure by industry, but instead would involve public expenditure to replace lead water distribution pipes.

The effectiveness measure developed in this analysis directly compares the effect of the two different regulations on changes in blood lead distributions. Since it is likely that similar changes in blood lead levels due to airborne and waterborne lead result in the same degree of health impact on the exposed population, this measure of effectiveness is the next best thing to having a direct measure of health impact. Direct comparisons of changes in the population at risk of the various lead-induced diseases are easily made, and such an approach could theoretically be applied to any regulation (e.g. the section 211 regulation limiting lead levels in gasoline) which affects lead exposure in the general population.

A.5.3 Feasibility of Cost and Effectiveness Comparisons Between the Three Lead Regulations and Other Environmental Regulations

There are no special features of these three regulations governing exposure to lead that would make cost comparison between them and other environmental regulations impossible. It is doubtful, however that the effectiveness of the lead regulations in reducing risks to public health could be directly compared to that produced by other regulations, except rather crudely. For example, the reduction in populations at risk of lead-induced disease might be compared to similar reductions produced by other regulations, but since there are no quantitative measures of the degree to which the lead regulations actually reduce the incidence of disease, comparisons of these regulations with other regulations along the commonly-used dimensions of health impact (deaths prevented, life-years saved, lost activity, etc.) are not possible. This difficulty is exacerbated by the fact that many of the health effects of lead exposure, such as the early stages of central nervous system damage, are subtle and difficult to quantify.

APPENDIX B

CASE STUDY OF THE NATIONAL EMISSIONS STANDARD FOR VINYL CHLORIDE

APPENDIX B

CASE STUDY OF THE NATIONAL EMISSIONS STANDARD FOR VINYL CHLORIDE

B.1 INTRODUCTION

The potential for exposures to vinyl chloride increased rapidly with the widespread use of vinyl chloride both as a gas and as the monomer of polyvinyl chloride (PVC). PVC, a plastic, is used extensively in the construction industry, as a packaging material and in consumer products.

Vinyl chloride has been implicated as a carcinogen in occupational health studies. In 1974, the B.F. Goodrich Company reported that several of its workers in a plant that polymerizes vinyl chloride into polyvinyl chloride had died of liver angiosarcoma, which is a very rare cancer in the general population. Other companies have since reported the incidence of angiosarcomas among workers exposed to vinyl chloride; the rates are estimated to be from 400-3,000 times the expected rate in the general population. Other effects such as nonmalignant liver damage and a group of symptoms classified as "vinyl chloride disease" (Reynaud's phenomenon, decreased circulation in the extremities, and acro-osteolyses -- a degeneration of the bones in the fingers and toes) were reported. OSHA set an emergency temporary standard for vinyl chloride in the workplace at 50 ppm. As evidence developed about the risk of cancer at lower levels of exposure (inhalation at 10 ppm) OSHA issued its new standard limiting exposure to no more than 1 ppm per 8 hour workshift.

Although the highest levels of exposure to vinyl chloride are found in the workplace, vinyl chloride is also found in the air, water and in food. An important source of exposure for the general population was in emissions from plants which either manufacture vinyl chloride from petrochemicals, polymerize the gas into the solid plastic, or fabricate the plastic into products for consumer use or use in other industries.

EPA promulgated a regulation for vinyl chloride under Section 112 of the Clean Air Act. This section of the Act allows EPA to designate chemicals "anticipated to result in an increase in mortality or an increase in serious irreversible or incapacitating reversible illness" as hazardous air pollutants and to set emissions standards to protect the public health "with an ample margin of safety." Vinyl chloride was listed as a hazardous air pollutant in December 1975 and the regulation setting the limit of 10 ppm was published in October 1976.

The vinyl chloride regulation was selected for this feasibility analysis as a representative regulation under Section 112. There are a total of five chemicals listed as hazardous air pollutants and regulated under this section: asbestos, benzene (proposed), beryllium, mercury and vinyl chloride. In the case of vinyl chloride, EPA had already performed a risk reduction analysis to determine how many people would be protected from contracting liver cancer by the EPA standard. There were extensive data available from occupational exposure studies and dispersion models. Both asbestos and benzene have been widely studied as workplace hazards and the health effects data available are of a similar type. The study of the effectiveness would be carried out in the same way that we studied vinyl chloride. There are less data available for beryllium and mercury; thus, smiliar analyses for these substances would be more difficult. EPA has not identified the population at risk for asbestos, beryllium and mercury.

B.2 ASSESSMENT OF RISK REDUCTION

In order to measure risk reduction for the vinyl chloride regulation, it is necessary to know the risk at different exposures to vinyl chloride, the different levels of exposure which existed before the regulation and which were brought about by the regulation, and the number of people exposed. EPA was able to predict all of these for vinyl chloride,[1] and estimated that one of the benefits of the vinyl chloride regulation was a 95% reduction in the number of cases of cancer, from a maximum of 20 per year to 1 per year, in the areas where vinyl chloride was manufactured. This section presents the analysis of the risk reduction calculation.

B.2.1. Estimates of Risk

We have some information on human occupational exposure to vinyl chloride, but we do not have information on the levels of exposure. Thus, we are required to use animal data in predicting the response at different levels.

a. Estimates from Animal Data

Dose response data were available from inhalation experiments with rats exposed to various concentrations of vinyl chloride daily over a period of one year, or approximately one half the average lifetime of a rat. The experimental animals showed an increased number of liver angiosarcomas and other cancers (Exhibit B-1). Since the dosages used in the laboratory were higher than those usually found in the environment, estimates were made for response at lower levels, using the data for exposure below 500 ppm. EPA assumed a linear model to make these estimates.[2]

$$\text{Probability of angiosarcomas} = 2.53 \times 10^{-4}d$$

$$\text{Probability of any tumor} = 0.123 + 5.26 \times 10^{-4}d$$

where d = dose in ppm.

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[1] Kuzmack, A. M. and R. E. McGaughy, Quantitative Risk Assessment for Community Exposure to Vinyl Chloride. EPA, December 5, 1975.

[2] A more correct procedure may have been to fit the animal data to a multistage model as recommended by EPA's Carcinogen Assessment Group. However, the error introduced into the analysis by this procedure is probably small, because at low doses the multistage exponential model is almost equivalent to a linear model.

EXHIBIT B-1

LIFETIME TUMOR INCIDENCE IN RATS EXPOSED TO VINYL CHLORIDE

(Inhalation 4 hr/day, 5 days/week for 52 weeks)

VC Concentration (ppm)	Number of Liver Angiosarcomas	Number of Tumors of all Types
10,000	9/61 (15%)	38/61 (62%)
6,000	13/60 (22%)	31/60 (52%)
2,500	13/59 (22%)	32/59 (54%)
500	7/59 (12%)	22/59 (37%)
250	4/59 (7%)	16/59 (27%)
50	1/59 (2%)	10/59 (17%)
0	0/58 (0%)	6/58 (10%)

Since the rats were exposed for only 4 hours a day, 5 days a week, an adjustment has to be made for environmental exposure. EPA assumed that environmental exposure would be continuous for 24 hr/day, 7 days/week,[3] and corrected for the difference in exposure by dividing the continuous dose received by people by (7x24)/(5x4)=8.4. This assumes that an intermittent dose is equivalent to a continuous dose of the same size, which is consistent with the model developed by EPA's Carcinogen Assessment Group.[4]

The annual risk to rats of contracting cancer from vinyl chloride exposure was developed in terms of the daily dosage in parts per million. EPA then extrapolated this data to human risk by making the assumption that the same number of cancers would be induced in a human lifetime as would be induced in the lifetime of a rat with similar exposure. The equivalent length of exposure in humans would be approximately 30 years.

Since this risk analysis was published in 1975, EPA's Cancer Assessment Group (CAG) has published its recommendations for risk calculation for carcinogens.[5] The same data can be used, but should use the CAG assumptions:

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[3] This does not account for the daily mobility of the population in and out of the exposure area.

[4] Cancer Assessment Group's Method for Determining the Unit Risk Estimate for Air Pollutants, EPA, July 31, 1980.

[5] Kuzmack, A. M. and R. E. McGaughy, Quantitative Risk Assessment for Community Exposure to Vinyl Chloride, EPA, December 5, 1975.

1) fit the dose/response data to a multistage model

2) extrapolate animal risk to human risk using the dose per relative surface area.

As we discussed in Chapter 2, by making these extrapolations, we assume that the mechanism by which vinyl chloride acts is the same in humans and in animals, and that there is a relationship between surface area and dosage to produce an effect.

b. Use of Human Data

Human epidemiological data can be used to support the animal risk data; unfortunately these studies from the workplace do not have accurate measurements of vinyl chloride levels in the environment. Therefore, they cannot be used to develop a dose response curve for human exposures. In one case where exposures were known (120-385 ppm over 8 hr/day, 5 days/week with excursions up to 4,000 ppm) the sample size was too small to detect liver cancers, although cases of dizziness on the job were reported. As evidence of toxicity was discovered, conditions in the plants were improved. More typical data from studies of occupationally exposed populations are shown in Exhibit B-2.

EXHIBIT B-2

LIVER ANGIOSARCOMA INCIDENCE AMONG HIGHLY EXPOSED WORKERS

Number of Cases of Angiosarcoma	Number of People Surveyed	Incidence ($x 10^{-3}$)
6	1,817	3.3
3	151	20.0
7	270	26.0
6	745	8.0

The definition of high exposure and the duration of exposure varied from study to study and from plant to plant within the same study. In addition to the inaccuracy of the data, it is difficult to convert this data to environmental exposures which tend to be much lower.

B.2.2 Levels of Exposure to Vinyl Chloride

a. Baseline levels

EPA used the long-term average concentration of vinyl chloride as a measure of exposure, since there were no data available on the response to peak exposures. The data used were from diffusion modeling techniques which estimated the concentration of vinyl chloride at various distances within a 5 mile radius of PVC and vinyl chloride monomer (VCM) plants, considering the population distribution at various distances and variations in size of plants. The average exposure in this area was estimated to be 17 ppb. The calculations of risk reduction were based on the assumption that without a regulation, emissions would continue at this level.

b. Estimated Levels to be Achieved by Standard

EPA has imposed specific controls on the manufacturers and polymerizers of vinyl chloride which they estimate will reduce emissions, and exposures in the 5 mile radius of the plants, by 95 percent. This decrease in exposure was used in their risk reduction estimate.

EPA is currently reviewing the implementation and enforcement of the current vinyl chloride standard. A report on this study is due to be published April, 1981. Thus, there may soon be more recent data on the exposure levels of vinyl chloride.

B.2.3. Number of People Exposed

The information on the population exposed to environmental vinyl chloride comes from a study performed by the American Public Health Association, based on 1970 Census data. This showed the numbers of people located at various distances up to 5 miles from VC plants to be 4.6 million. This refers to the people living in the area and did not account for daily mobility in and out of the area.

B.2.4. Risk Reduction

From these estimates, EPA predicted that before the standard, the number of angiosarcoma cases each year expected to develop due to continuous exposure at current levels of vinyl chloride to be between one and ten; the number of other cancer cases per year was estimated to be in this same range. Therefore, vinyl chloride exposure was estimated to cause up to 20 cases of cancer per year. EPA estimated that its regulations would reduce exposure by 95%. Assuming a linear model, a 95% reduction in exposure would reduce the maximum numbers of cancers from 20 to one, and the maximum benefits per year would be a reduction of 19 cancers per year.

The maximum benefits would be expected in the population which had never been exposed at the higher levels. The dose response model predicts the incidence of cancer for the population exposed to vinyl chloride over approximately half a lifetime (or thirty years). From this, it is relatively easy to predict the incidence in the population exposed to vinyl chloride at the higher levels for a period of thirty yers, and in those exposed at the lower levels for thirty years. However, it is not clear how exposure at the reduced emissions would affect the part of the population that had received a few years exposure at high levels followed by exposure at the lower levels. The model does not predict the incidence in a population which was exposed to varying levels of vinyl chloride. Thus, we may predict the incidence of cancer in the thirtieth year after the emission standard is achieved, but do not have the data to directly predict the incidence before that time. To calculate the total number of cancers prevented over a period of several years right after the implementation of the standard requires an understanding of how total cumulative dose is related to cancer incidence. Assuming a linear model, with the one-hit theory of initiation of cancer, we could predict a gradually decreasing number of cases of cancer as a result of vinyl chloride emissions over a period of years until the maximum reduction was achieved at thirty years.

This points out the importance of being explicit about the time period over which benefits are measured. A statement that the new regulation reduces cancer incidence by 19 cases per year does not acknowledge that this reduction is to be achieved in the future. While this may not be a problem in the discussion of an individual regulation, it may complicate the comparison of regulations, where the benefits are realized in different time periods. This problem is discussed in Chapter 2.

B.2.5. Evaluation of Risk Reduction

This discussion was based on the EPA assessment of risk reduction which gives an estimate for the effectiveness of the regulation in preventing cancer in the population due to exposure to vinyl chloride. As pointed out above, the estimate was based on the 1970 population statistics and assumed that the emission levels would continue at the 1974 level. For an in-depth analysis of the effectiveness of the vinyl chloride regulation, it would be necessary to update this information. In addition, the conversion of dose response data should be recalculated by using the CAG methods, in order to make the assessment consistent with those for other regulations. Calculations of the expected benefits over the years after the standard is implemented up to the time when the maximum benefits are achieved should be made in order to make accurate estimates of the total number of cancers prevented by reducing vinyl chloride emissions.

Although there have been other health problems such as non-malignant liver damage associated with vinyl chloride exposure, EPA did not estimate how wide-spread these were in the general population and how the new regulation would affect the incidence. Most of these were associated with high occupational exposures and it would be difficult to predict their incidence at the lower exposures. More importantly, however, known effects of environmental exposure to vinyl chloride such as the production of increased incidence of birth defects in exposed populations were also not included in the risk assessment.

B.3 ASSESSMENT OF COST IMPACTS

B.3.1. Coverage

The standard applies to ethylenedichloride/vinyl chloride (EDC/VC) and PVC plants, which are responsible for 96 percent of the total nationwide vinyl chloride emissions. PVC fabrication plants were exempt since the vinyl chloride emissions around these facilites were negligible. In 1975, there were 17 EDC/VC and 41 PVC plants.

B.3.2 Impacts Projected by EPA

a. Cost

EPA estimate that the total costs for these plants were as follows (for air requirements only):

	$ (million) 1975	
Plant Type	Total Capital Costs	Total Annualized Costs
EDC/VC	15	12
PVC	183	58
TOTAL	198	70

Of these, the following are assumed to be costs which may be attributable to OSHA (or allocated jointly between EPA/OSHA):

EDC/VC	8	6
PVC	29	19
TOTAL	37	25

One possible error is the approach to define model plants for EDC/VC and PVC production. Unit costs for model plants do not appear to be sufficiently documented. In addition, methods for extrapolating model plant costs to industry are not justified (for example, different economies of scale were sometimes assumed for different controls, with no stated justification). The method for annualizing capital costs was the traditional annual capital charge (ACC) computed at (10%, N) where N varies for different types of control equipment. Also, an administrative charge of 0.025 was added to the ACC to cover property taxes, insurance, and overhead. This is not standard practice and adds 15%-20% to capital costs which is not justified. It may be reasonable to treat these costs as annual operating costs. Assuming the 0.025 factor is valid (it is not documented), correcting for this assumption would shift approximately 15-20% of the capital costs to operating costs.

Another error appears to be the omission of cost data for new plants. The cost data presented above apply only to existing plants as of 1975. At that time, there was some planned new capacity. It would be worthwhile to compare 1975 data with 1980 data on plants and capacity to determine if any coverage adjustments need to be made. Exhibit B-3 shows the numbers of EDC/VC and PVC plants and the capacity in pounds for 1974 and estimated for 1981. This shows a capacity increase of 50% for EDC/VC plants and 37% for PVC plants. These new plants will have to incur costs to comply with the regulation, which will increase the overall cost estimates. The new plants may incur lower compliance costs as the controls may be less expensive to design into the plants initially than to add-on to existing plants.

EXHIBIT B-3

CAPACITY COMPARISON

Year	EDC/VC Plants	Capacity
1975	17	6.8 billion lb
1980		8.2 billion lb + 2.0 billion new (G/P, PPG, Dow)

Capacity Increase 1975-81 = 50%

	PVC Plants	
1975	41	5.7 billion lb
1980		7.6 billion lb + .2 billion new

Capacity Increase 1975-81 = 37%

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*The data for 1975 are based on EPA estimates in Standard Support and Environmental Impact Statement: Emission Standard for Vinyl Chloride, p. 7-67. The data for 1980 are from Chemical and Engineering News [illegible]7/80), p. 9-13.

b. Economic

The economic analysis focused on two economic effects, price increases and plant closures. Below is a summary of projected effects due to VC air requirements:

Plant Type	Price Increase	Closure
EDC/VC (existing)	0.4%	0
PVC (existing)	7.3%	4 (0.5% of capacity)

B.3.3 Revised Impacts

a. Actual Data

It has been suggested that the actual costs of compliance for vinyl chloride were less than those initially estimated because of technological innovation. Areas where technological innovation may be significant include:

- improved stripping (will reduce costs for PVC plants)
- portable exhaust ventilation to deal with fugitive emissions (EDC/VC and PVC plants)

However, we can safely assume that these technological innovations result in the overstatement of unit costs, particularly for PVC plants. Whether or not they offset the coverage underestimates is a matter of conjecture.

Capacity has grown by 50% in VCM manufacturing and by 37% in PVC manufacturing. The relationship between cost and capacity is at most linear (due to economies of scale). Assuming a linear relationship as an upper bound, technological innovation would have to reduce unit costs by something approaching 50% in VCM manufacturing and something approaching 37% in PVC manufacturing in order for these two adjustments to offset each other. This may be reasonable for PVC, but in VCM manufacturing, total costs probably need to be adjusted upwards.

b. Calculation of Revised Impacts

1) Assume all costs for fugitive emissions are attributable to OSHA. The costs attributable to the EPA regulation are:

	1975 $ million	
	Total Capital	Total Annualized
EDC/VC	7	6
PVC	154	39
TOTAL	161	45

2) Reduce capital costs by 15% to account for misclassification of property taxes, insurance, overhead.

	1975 $ million	
	Total Capital	Total Annualized
EDC/VC	6	6
PVC	131	39
TOTAL	137	45

3) Approximate Capital/Annual by assuming the cost of capital to be 10% and the equipment life to be 15 years.

	1975 $ million			
	Total Capital	Annualized Capital	Operating and Maintenance	Total Annualized
EDC/VC	6	1	5	6
PVC	131	17	22	39
TOTAL	137	18	27	45

4) Adjust EDC/VC costs to account for coverage (50%).[6]

	1975 $ million			
	Total Capital	Annualized Capital	Operating and Maintenance	Total Annualized
EDC/VC	9	1	8	9
PVC	131	17	22	39
TOTAL	140	18	30	48

5) Convert to 1979 dollars (1975 GNP deflator = 127.15, 1979 GNP deflator = 165.50) (increase costs by 165.5/127.15 = 30.2%).

	1979 $ million			
	Total Capital	Annualized Capital	Operating and Maintenance	Total Annualized
EDC	12	2	10	12
PVC	171	22	29	51
TOTAL	183	24	39	63

B.4 COST EFFECTIVENESS OF THE VINYL CHLORIDE NATIONAL EMISSIONS STANDARD

EPA estimated that the reduction in emissions brought about by the vinyl chloride standard would be a reduction of from one to 19 cases of cancer. We have pointed out that this reduction would be expected in approximately thirty years after implementation of the standard. Although there is reason to expect that there will be a reduction in cancers due to vinyl chloride before this time, this has not been calculated and is not included in the discussion. Thus, this discussion may understate the benefits.

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[6] The actual adjustment would, no doubt, be less than 50% because compliance costs for new capacity should be less than for existing plants. A more accurate adjustment could be made after review of actual costs or thorough engineering analysis.

There are problems with estimating the cost of the regulation over the next thirty years. Predicting what the emissions would be with and without the regulation becomes difficult that far in the future. The capital equipment life is assumed to be fifteen years. After fifteen years, investment may be made in new control methods, but the cost is difficult to predict. This makes the cost estimates very uncertain.

The capital costs are $183 million, with yearly operating and maintenance costs of $39 million. At a 10 percent cost of capital, the present value of these costs is $653 million. The benefits to be derived from this investment are assumed to begin thirty years from the time the exposure levels are reduced. If we assume that the emission levels continue at the low rate, this thirty years investment should affect the exposed population over the next thirty years. The maximum total cancers to be prevented by the implementation of this standard over thirty years is 570. The cost per cancer prevented is at least $1.15 million. Since this calculation assumes the maximum benefits, this number is a lower bound.

APPENDIX C

CASE STUDY OF THE CHLOROFLUOROCARBON AEROSOL PROPELLANT BAN*

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*Note: This case study draws heavily on "Regulation of Nonaerosol Uses of Chlorofluorocarbons," by Michael Gibbs, April 1980. That paper was written while Mr. Gibbs was a student at the John F. Kennedy School of Government. Mr. Gibbs is now with ICF Incorporated.

APPENDIX C

CASE STUDY OF THE CHLOROFLUOROCARBON AEROSOL PROPELLANT BAN

C.1 INTRODUCTION

In 1976 EPA, the Food and Drug Administration (FDA), and the Consumer Products Safety Commission (CPSC) announced their intention of initiating the regulatory process for CFCs. A regulatory plan was developed that divided the regulation into two phases: Phase I - nonessential aerosol propellant uses; and Phase II - all other uses. FDA had authority over 85 percent of the Phase I uses of CFCs. With the passage of the Toxic Substances Control Act (TSCA), EPA and CPSC had duplicate authority over the remaining 15 percent. Seeking to avoid regulatory duplication, CPSC allowed EPA to take the lead in regulating products coming under the jurisdiction of both agencies. EPA and FDA cooperated in banning the use of CFCs as a nonessential aerosol propellant.

The ban on the production of CFCs as a propellant took effect October 15, 1978 and as of April 1979, nonessential aerosols containing CFC propellants could not be sold in interstate commerce. Technically the Phase I rule only covers "fully halogenated chlorofluoroalkanes." This generic category includes CFC-11, CFC-12, CFC-113, CFC-114, and CFC-115, but does not include other CFCs which at the time were believed to be potential substitutes for those banned. Aerosol products using CFCs as both a propellant and an active ingredient, such as a stabilizer, were not covered in the ban and will be treated, if needed, under the Phase II regulation.

The Phase I rule also included an exemption clause. Exemptions are granted on a case by case basis to specific aerosol propellant uses if it is proved that that use is "essential." The requirements for proof of essentiality are open to interpretation, however, only about two percent of all aerosol propellant uses are expected to be exempted. Some products that have been granted exemptions are:

- warning devices for mines;
- release agents for molds used in the production of plastic and elastomeric materials;
- flying insect pesticides used in nonresidential food handling areas and for space spraying of aircraft;
- diamond grit spray; and
- some pharmaceutical uses.

All the exempted uses will be examined under Phase II.

This case study first examines the current state of the art in assessing the propensity of CFCs to deplete stratospheric ozone. The adverse effects that increased exposure to ultraviolet radiation resulting from ozone depletion can have on humans and the environment are then presented and quantified to the extent possible. The costs of the CFC aerosol propellant ban are discussed, and finally a range of the cost-effectiveness of the regulation is developed.

C.2 ASSESSMENT OF RISK REDUCTION

C.2.1 Introduction

There are six commercially produced chlorofluorocarbons (CFCs) widely used in the United States. In the concentrations that are normally found, there are no adverse health effects from direct contact with CFCs. Consequently, CFCs have replaced other more toxic chemicals in several uses.

Through a complex process, five of the six commercially produced CFCs are believed to reduce the steady state concentration of ozone in the stratosphere. Ozone is important because it shields the earth's surface from harmful ultraviolet rays. For eample, a reduction of stratospheric ozone of 16.5 percent is predicted to result in a 33 percent increase in harmful ultraviolet radiation reaching the earth's surface.

There are no beneficial effects of this increase in ultraviolet radiation. The increase in ultraviolet radiation may harm plants and animals but there is insufficient information to estimate the effects. The increase in ultraviolet radiation will increase the incidence of melanoma and nonmelanoma skin cancers in humans. A 16.5 percent reduction in the concentration of stratospheric ozone could cause an increase of approximately 66 percent in the incidence of each of the two types of skin cancer. The benefit of the CFC aerosol propellant ban is the prevention of an increase in the incidence of these cancers.

C.2.2 Background

What are CFCs?

Chlorofluorocarbons (CFCs) are clear, highly volatile fluids which have a high vapor pressure, high density, and low boiling point.[1/] They were invented in the early 1930s at the General Motors Research Laboratory. Most CFCs are comprised exclusively of carbon, chlorine and fluorine and are called "fully halogenated" chlorofluorcarbons because they contain no hydrogen. There is one non-fully halogenated CFC that is widely used today, CFC-22. It accounts for approximately twenty percent of CFC production in the U.S.[2/] However, the ability of CFC-22 to adversely affect stratospheric ozone is unknown at this time. Its use has not been banned and it is not included in this case study.

[1/] Final EIS, Fluorcarbons: Environmental and Health Implications, FDA, February 1978, page 7.

[2/] Emissions Control Options for the Synthetic Organic Chemicals Manufacturing Industry, Hydroscience, EPA Contract No. 68-02-2577, June 1979, page II-2.

The five fully halogenated CFCs that are manufactured in commercial quantities in the U.S. are:[3]

- CFC-11 (CCl_3F);
- CFC-12 (CCl_2F_2);
- CFC-113 ($C_2Cl_3F_3$);
- CFC-114 ($C_2Cl_2F_4$); and
- CFC-115 (C_2ClF_5).

CFC-11 and CFC-12 were the two primary propellants affected by the ban, although the other CFCs were used as propellants in a few applications.

CFCs were used to replace ammonia, sulfur dioxide, and ethylene as the working fluid (refrigerant) in refrigerators, freezers, and air conditioners. CFCs were preferred because of their low toxicity and nonflammability.[4] After low-pressure valves were developed in the 1950's, CFCs became the standard aerosol propellant in the U.S. Its major advantage was its ability to maintain an even spray with very small particle size and even dispersion. The first application of CFCs as an aerosol propellant was to spray insecticide. During the 1960s and 1970s its use as a propellant in personal convenience products grew rapidly until 1978 when this use was banned.[5] The ban is expected to reduce U.S. emissions of CFCs by approximately fifty percent of 1977 levels. CFCs are still permitted to be used as aerosol propellants in most countries throughout the world. The major nonaerosol applications of CFCs in the U.S. are: as a working fluid (refrigerant) in refrigeration and air conditioning equipment; as a blowing agent and insulating gas for pastic foams; as a solvent for cleaning and drying; and as a diluent gas in sterilization processes.

Ozone Depletion Theory

Ozone (O_3) is a trace constituent of the stratosphere. Although ozone represents only a few parts per million of the gases in the stratosphere, it plays a vital role in shielding the earth from potentially damaging rays from the sun. Threats to the so-called "ozone layer" in the stratosphere have become a public concern in recent years and it is currently believed that CFCs pose the most serious threat.

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[3] The numbering system for CFCs is as follows: CFC-ABCD

D = No. of fluorine atoms
C = No. of hydrogen atoms plus 1
B = No. of carbon atoms minus 1
A = No. of double bonds.

[4] *Final EIS* ..., *op. cit.*

[5] "The Fluorocarbon-Ozone Theory - I," by R. L. McCarthy, F. A. Bower, and J. P. Jesson, *Atmospheric Environment* 11:6 (1977), page 491.

A simple understanding of the dynamics of the atmosphere is required to understand the main points of the ozone depletion theory. (See Exhibit C-1). The troposphere is the lowest portion of the atmosphere, 0-8 km near the poles and 0-16 km near the equator. The troposphere is subject to violent mixing, what we experience as weather. The temperature of the air in the troposphere decreases with altitude until the minimum temperature (approximately -80°F) is reached at the tropopause.

The stratosphere begins at the tropopause and continues up to the stratopause at approximately 50 km. In the stratosphere the air temperature increases from the -80°F at the tropopause to a maximum of approximately +50°F at the stratopause.6/ The stratosphere is stable and virtually cloudless. There is very little mixing because the denser, cooler air at the lower altitudes does not readily rise. Gases in the stratosphere are primarily mixed by large-scale, nearly horizontal movements of air masses.7/ The relative quiescience of the stratosphere makes it particularly susceptible to contamination because pollutants introduced there tend to remain for long periods of time.8/

The equilibrium concentration of ozone in the stratosphere is the result of a dynamic balance between formation and destruction processes. Ozone is formed in the stratosphere primarily through the photodissociation of molecular oxygen resulting in oxygen atoms that rapidly combine with molecular oxygen in the presence of another molecule.9/ The following equations describe this formation process:

O: atomic oxygen

O_2: molecular oxygen

O_3: ozone (odd oxygen)

M: another molecule

$h\nu$: ultraviolet radiation below 242 nm

$$O_2 + h\nu \rightarrow O + O$$

$$O + O_2 + M \rightarrow O_3 + M \text{ (twice)}$$

$$\text{Net:} \quad 3O_2 \rightarrow 2O_3$$

6/Halocarbons, Effects on Stratospheric Ozone, National Research Council, National Academy of Sciences, 1976, page 21.

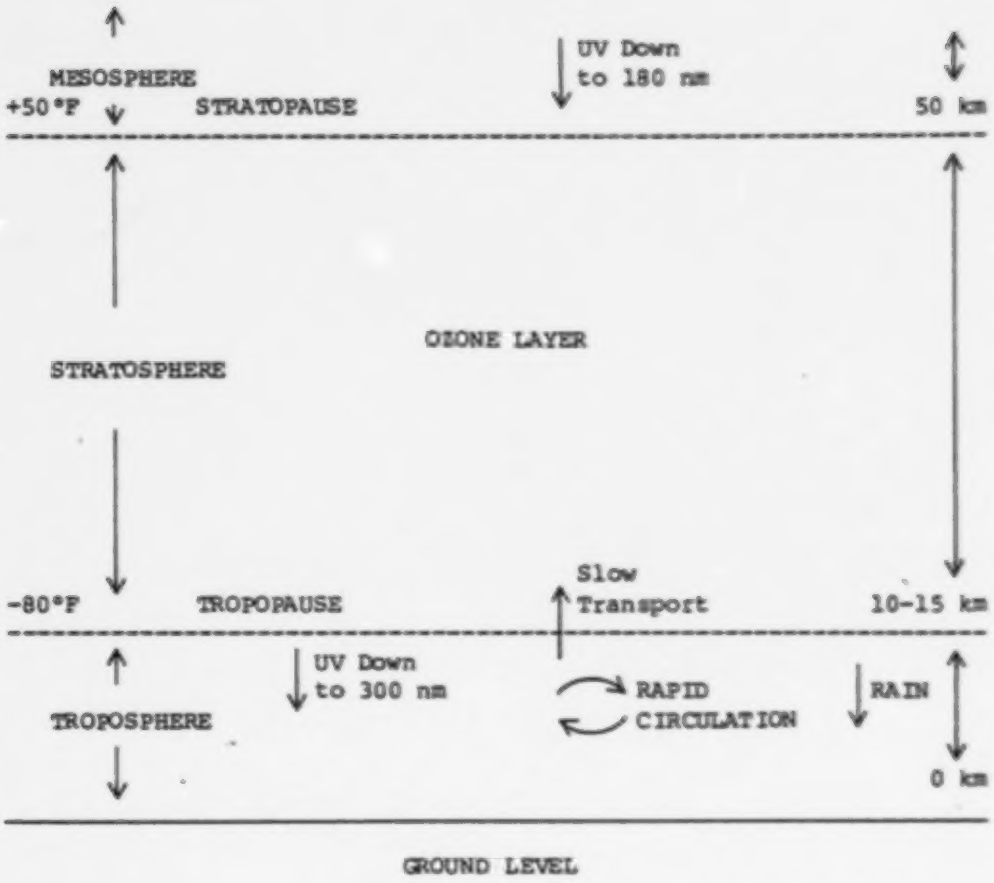
7/Ibid., page 22.

8/Ibid., page 1.

9/Ibid., page 23.

EXHIBIT C-1

THE ATMOSPHERE



The formation process is balanced by three naturally occurring destruction processes. The first process absorbs ultraviolet radiation in the wavelength range of 240 nm to 320 nm and is the important ozone destruction process that protects the earth's surface from the potentially harmful solar radiation in this wavelength range,

$$O_3 + hv \longrightarrow O_2 + O \qquad \text{where } 240 \text{ nm} < hv < 320 \text{ nm}.$$

This accounts for approximately twenty percent of the total natural destruction rate for stratospheric ozone.[10]

About ten percent of the destruction of ozone is caused by catalytic cycles involving hydrogen containing compounds. The remaining seventy percent of the natural ozone destruction rate is believed to be caused by small amounts of NO_x compounds or nitrogen oxides. NO_x originates in the troposphere where almost all of it is converted to N_2 and O. Only about one percent of it reaches the stratosphere to react with ozone. The NO_x molecules that do reach the stratosphere are removed by the formation of nitric acid (HNO_3) in the lower stratosphere. HNO_3 is stable and does not react with ozone. It is eventually transported down across the tropopause into the troposphere and rained out.[11]

The introduction of CFCs into the troposphere is believed to significantly alter the equilibrium of these natural processes. In 1974, F. S. Rowland and M. J. Molina put forth their "Ozone Depletion Theory" that demonstrated the potential of CFCs to adversely affect the ozone layer.[12] CFCs are released into the troposphere and do not breakdown because they are very stable.[13] They are slowly transported across the tropopause into the stratosphere as part of a very complex and little understood mixing phenomenon between the stratosphere and the troposphere. The CFCs slowly rise upward toward the 25 km to 50 km region of the stratosphere. In this area they photodissociate by ultraviolet radiation of approximate wave length 200 nm and release their chlorine atoms. The CFCs can not photodissociate before reaching the upper stratosphere because ultraviolet radiation of wavelengths around 200 nm are screened out by the ozone formation process described above.

The chlorine atoms released from the CFCs in the upper stratosphere catalyze the destruction of ozone as described by the following cycle:

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[10] Ibid., page 24.

[11] Ibid., page 26.

[12] For details of the original theory see: "Stratospheric Sink for Chlorofluoromethanes: Chlorine Atom - Catalyzed Destruction of Ozone," Nature 249 (1974).

[13] This applies to the fully halogenated CFCs only.

$$\begin{array}{lrcl} & Cl + O_3 & \rightarrow & ClO + O_2 \\ & ClO + O & \rightarrow & Cl + O_2 \\ \hline \text{Net:} & O_3 + O & \rightarrow & O_2 + O_2 \end{array}$$

The addition of this destruction process reduces the equilibrium concentration of ozone in the stratosphere. Therefore, there is less ozone to be destroyed by absorbing radiation in the wavelength range of 240 nm to 320 nm and more of this radiation will reach the earth's surface. The resultant increase in ultraviolet radiation depends on how significantly the equilibrium level of ozone is reduced.

The chlorine atoms released from CFCs in the upper atmosphere can destroy many ozone molecules before the catalytic cycle is interrupted by conversion of the highly reactive Cl and ClO compounds into inactive forms (such as HCl and $ClONO_2$) that do not deplete ozone. These relatively inactive forms are removed from the stratosphere primarily through mixing with the upper troposphere after which they are washed out by rain. It is hypothesized to take several decades for the chlorine in ground level CFC emissions to return to earth via rain after having been in the upper stratosphere. Therefore, the full effects that the tremendous growth in the use of CFCs in the past twenty years may have on stratospheric ozone have yet to be felt.[14] Since stratospheric production of ozone is relatively insensitive to man's activities,[15] any catalytic destruction of ozone by the chlorine from CFCs would eventually be reversed if CFC emissions were stopped. The reversal would take a long time because, as mentioned above, chlorine is removed from the stratosphere very slowly.

If CFC emissions were continued at constant annual rates, a new equilibrium concentration of stratospheric ozone would result. The difference between the natural equilibrium concentration of stratospheric ozone and the new equilibrium concentration due to the addition of the CFC induced chlorine destruction processes is called ozone depletion. Therefore, fifteen percent ozone depletion would mean an equilibrium concentration of ozone fifteen percent less than the natural equilibrium.

Ozone depletion is estimated by modeling the entire process described above with the aid of computers. Approximately twenty to thirty chemicals, one hundred reaction rates and one hundred reaction products are represented mathematically by a series of functions called "equations of mass conservation." The composition, temperature, and pressure of the stratosphere vary with altitude, latitude, and to a lesser extent, longitude so that a complete solution of the mass conservation equations requires all three dimensions as variables.[16] Time is also an important variable because the reaction kinetics are believed to vary with time of day, time of year, and the eleven year sun spot cycle.

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[14] Halocarbons, Effects On ..., op. cit., page 29.

[15] Ibid., page 2.

The overwhelming complexity of the model and the unavailability of atmospheric concentration measurements at various altitudes on a global basis have necessitated significant simplifications in the model. One dimensional approximations have been used to estimate ozone depletion. The longitudinal and latitudinal dependencies have been averaged out, leaving altitude as the sole spatial variable.[17] The results of the one dimensional models, as they are called, must be interpreted as global averages of ozone depletion.[18]

Due to the violent mixing in the troposphere, isolated CFC emission will deplete ozone all around the world. However, the global average of ozone depletion may not represent the actual depletion in a specific location. The percent depletion at the equator is expected to be between 1.5 to 2.0 times the percent depletion at the poles. This is because, as shown in Exhibit C-2, the concentration of ozone varies significantly with latitude.

Therefore, even though all countries will experience the effects of CFC induced ozone depletion, those countries nearest the equator will experience the greatest amount of ozone depletion.

Some recent one-dimensional model predictions of ozone depletion are:

	Lower Bound	Average	Upper Bound
1975 IMOS Interagency Task Force	--	7%	--
1976 NAS Report	2%	7.5%	20%
1977 NASA Report	10.8%	--	16.5%
1979 NASA Report	--	15%-18%	--
1979 NAS Report	5%	16.5%	28%

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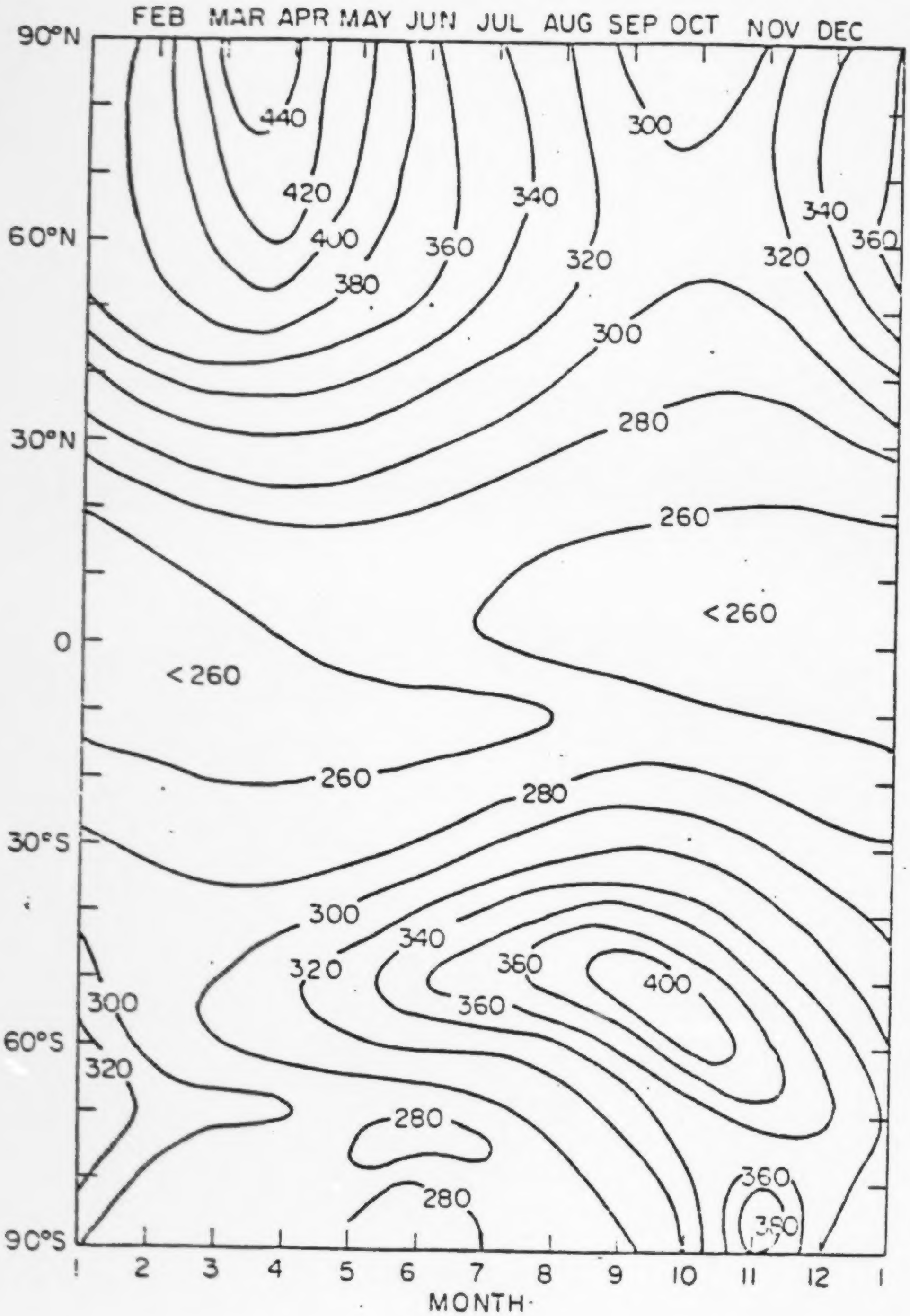
[16] Ibid., page 30.

[17] This includes averaging the concentrations, motions, and reactions over latitude and longitude leaving only their dependencies on altitude and time. See Ibid., page 6.

[18] Two-dimensional models are being developed to include latitudinal differences. Longitudinal variances are expected to be small.

EXHIBIT C-2

OZONE ABUNDANCE AS A FUNCTION OF LATITUDE AND TIME OF YEAR



The observed behavior of columnar ozone (Dobson units) as a function of latitude and season (Dütsch, 1971).

Source: The Stratosphere: Present and Future, page 246

There are many uncertainties that affect predictions of ozone depletion. Some uncertainties are exogenous to the model itself; for example the level of future CFC emissions. The above predictions assume CFC emissions constant at 1975 or 1977 rates (they are approximately equal) into the indefinite future. If emissions continue to increase as they have been, ozone depletion could be much greater than the above predictions. Exhibit C-3 shows time dependent predictions of ozone depletion for three emission scenarios. Some of the other major uncertainties are:

a. Tropospheric lifetime. Since CFCs are so stable, it has been assumed that they do not break down in the troposphere, but only break down in the presence of short wave ultraviolet radiation that reaches the stratosphere but does not reach the troposphere. If CFCs do break down in the troposphere, as indicated by some preliminary results of tests conducted by the National Bureau of Standards, then the effects of CFCs on stratospheric ozone may be much smaller than currently believed.

b. Transport kinetics. The mixing phenomenon between the troposphere and the stratosphere is not well understood. If more free chlorine radicals are passed down from the stratosphere at the same time that CFCs are passed up than is currently assumed, then the effects of CFCs may be slightly smaller than currently believed.

c. Atmospheric chemistry. There may be atmospheric chemistry missing from the theory. In the past several years the reaction kinetics, including reaction rates and reaction products, have been revised. The result of the revisions has been an increase in the predicted amounts of ozone depletion due to CFCs.

d. Effects of CFC types. There is considerable uncertainty as to the relative ability of other CFCs to deplete the ozone. It is believed that ozone depleting potential may be a function of the percentage, by weight, of chlorine in the molecule. Based on this assumption, the ozone depleting ability per pound of a CFC type, in relation to the ozone depleting ability of CFC-11 is:

CFC-11 : 1.0

CFC-12 : 0.76

CFC-113: 0.73

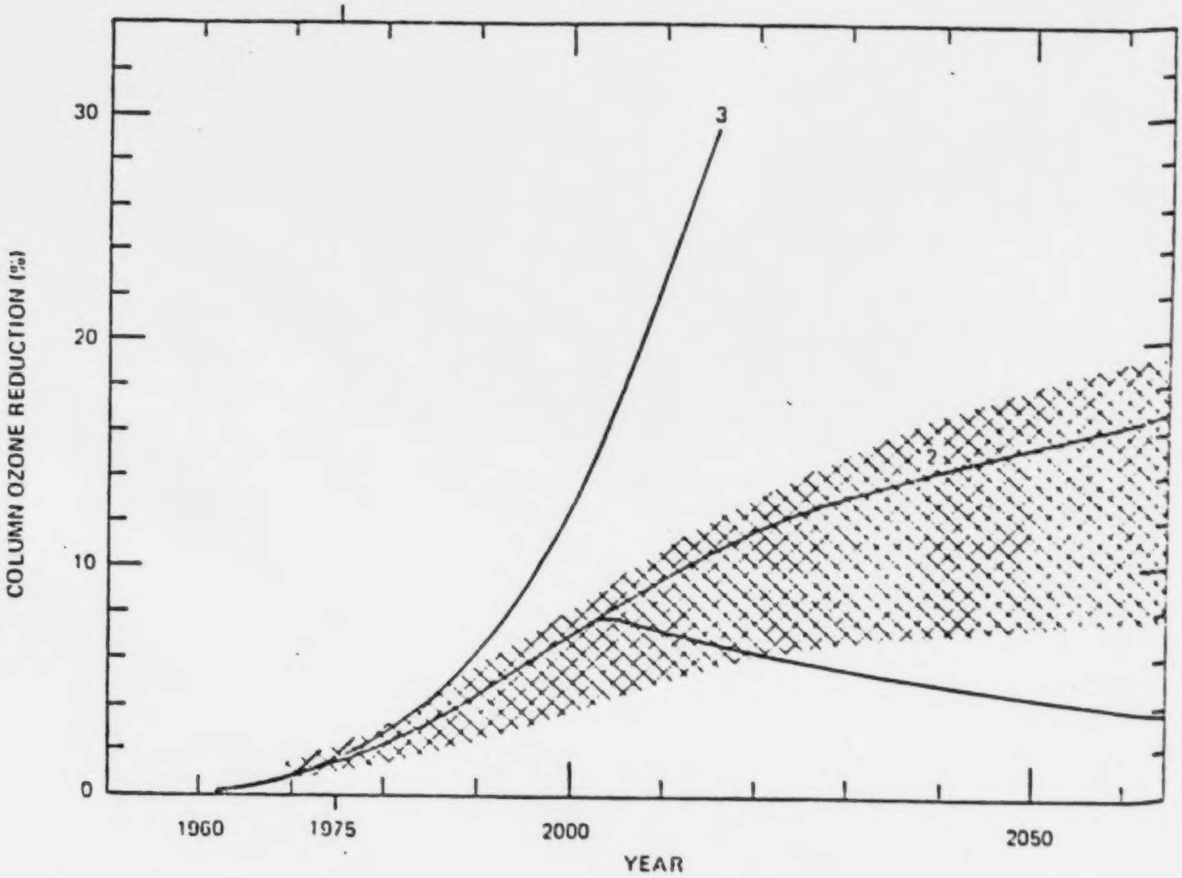
CFC-114: 0.54

CFC-115: 0.30.

However, characteristics such as molecular bond strength may also affect ozone depleting potential. The relative ozone depleting potential of the various CFCs remains uncertain at this time.

EXHIBIT C-3

MODEL PREDICTIONS OF TIME DEPENDENT OZONE DEPLETIONS



Typical model predictions of time-dependent ozone depletions for three fluorocarbon production (release) scenarios beyond 1975. They are (1) constant FC-11 and FC-12 release at 1975 rates until the year 2,000, with __no__ subsequent release; (2) constant release at 1975 rates indefinitely; and (3) release starting at the 1975 rates and growing by 5% per year for several decades. The tropospheric lifetimes of fluorocarbons 11 and 12 are assumed to be greater than 100 years. The fluorocarbon ozone reduction in 1975 is taken to be 1.5%. For curve 2, the cross hatched area reflects the widest range of model predictions available for this assessment.

Source: __The Stratosphere: Present and Future__, page 342.

Large amounts of empirical data are needed to validate or disprove the ozone depletion theory. The United Nations Environmental Programme (UNEP) has held several workshops to help coordinate CFC/ozone related research worldwide.[19] The following are highlights of some of the major research efforts.

The best way to firmly establish whether ozone depletion due to man's activities is actually taking place as expected is to match the results of time series ozone concentration trends with the model predictions.[20] Ground-based measurements of stratospheric ozone have been reported on a global basis since 1958 by the Canadian Atmospheric Service.[21] Balloon measurements have been made recently.[22] Also, NASA is currently in the midst of an ambitious satellite program that includes ozone detection. The Nimbus-7 Satellite is equipped with the Total Ozone Mapping System (TOMS) that, with the help of ground level observation calibration, will add significant new observation data.[23] Exhibit C-4 lists NASA's satellite sensors for ozone data.

However, due to daily and seasonal variations in ozone concentrations, individual measurements are of limited value. Only in the context of long-term trend analyses are the data valuable. To date, the difficulty in obtaining meaningful trend analysis results has been primarily due to the poor quality of the available data. Intercomparison of measurement instruments in 1972 revealed calibration differences of up to twenty percent. Current measurement techniques, in particular NASA's satellite program, are expected to improve detection accuracy to a range of ±5 percent.

But, even with this improved accuracy, small changes in ozone concentration will not be discernable from random measurement errors. If CFC emissions continue at current levels, it will be at least a decade, and possibly longer, before expected reductions in stratospheric ozone due to CFC emissions can be substantiated statistically with a high degree of confidence. Therefore, since there may be a significant amount of ozone depletion even if all CFC emissions were stopped in the year 2000, (see Exhibit C-3), the time series ozone concentration trend analyses are probably not an adequate "early warning system" upon which to base policy decisions.

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[19] Environmental Programmes of Intergovernmental Organizations, by Dr. P. L. DeReeder, The Netherlands, 1977.

[20] "Nonaerosal Propellant Uses of Fully Halogenated Halocarbons," Submission Number 2 by E. I. duPont de Nemours and Co., Inc., to EPA, June 1, 1979, page 8.

[21] Chlorofluoromethanes and the Stratosphere, NASA RP-1010 August 1977, page 67.

[22] Halocarbons, Effects On ..., op. cit., page 9.

[23] The Stratosphere: Present and Future, NASA RP-1049, December 1979, page 214.

EXHIBIT C-4

SATELLITE SENSORS FOR OZONE DATA

Satellite and Instrument	Ozone Measurement	Data Record	Comments
OGO-4 BUV	Profile	Sep 67 - Jan 69	Solar spectral irradiance inferred from rocket ozonesonde for OGO-4 calibration. Ozone analysis not completed.
Nimbus-3 IRIS	Total	Apr - July 69	Short data set, analysis complete, poor quality data.
Nimbus-4 BUV	Profile & Total	Apr 70 - July 77	Solar spectral irradiance assumed invariant beyond 1970. Total O_3 analysis completed. Profile analysis completed 1979. Validation with rockets and Dobson network.
Nimbus-4 IRIS	Total	Apr 70 - Jan 71	10 month's data, analysis complete.
Nimbus-6 LRIR	Profile	June 75 - Jan 76	Validation with rocket ozone profiles, day and night. Analysis not complete.
AE-5 BUV	Profile & Total	Nov 75 - May 77	Limited validation, preliminary analysis only, tropical latitudes.
Block 5 MFR series of 4 flights	Total (spatial scanning)	Mar 77 - July 77 July 77 - - - - - July 78 - - - - - June 79 - - - - -	O_3 from regression against climatology. Validation with Dobson network. Ozone analysis not complete.
Nimbus- SBUV	Total & Profile (Nadir)	Nov 78 - - - - -	Validation by rockets, balloons and Dobson network. Analysis started.
Nimbus-7 TOMS	Total (spatial scanning)	Nov 78 - - - - -	Validation by Dobson network Analysis started.
Nimbus-7 LIMS	Profile	Oct 78 - May 79	Validation by rockets and balloons. Analysis started.
TIROS-N HIRS	Total (spatial scanning)	Nov 78 - - - - -	Regression against Dobson stations. Analysis started.
Applications Explorer Mission - SAGE	Profile	Jan 79 - - - - -	Solar occulation method, validation with rockets, limited daily coverage of the globe. Analysis started.

Source: The Stratosphere: Present and Future, page 283.

To further the understanding of the atmospheric chemistry that leads to ozone depletion, Professor James G. Anderson of Harvard University has made several in situ measurements of various stratospheric constituents. Most notably he has measured ClO and O_3 simultaneously. The results of his experiments showed unexpectedly high concentrations of ClO with normal concentrations of ozone.[24] This is significant because, as mentioned above, ClO is believed to be the primary chlorine species that causes the catalytic destruction of ozone. If confirmed, Professor Anderson's results imply that there are large unknown sources of chlorine that reach the stratosphere and that ClO may be more stable in the presence of ozone than is currently believed. Professor Anderson is repeating and expanding his measurements, and at least two other groups are studying ClO concentrations by different measurement techniques.[25]

Another important uncertainty in the ozone depletion theory that is being investigated is the magnitude of CFCs' tropospheric lifetime. To date, all models have incorporated the assumption that CFCs do not break down in the troposphere; i.e., have an infinite tropospheric lifetime. Estimates of ozone depletion are highly sensitive to this assumption. For example, if CFCs' true tropospheric lifetime was ten years, ozone depletion estimates would be reduced by a factor of ten.[26]

A detailed materials balance is needed to determine the tropospheric lifetime of CFCs. This entails comparing the quantity of CFCs that has been released into the atmosphere to the sum of the quantity currently in the troposphere plus the quantity estimated to have been used up in the stratospheric process that destroys ozone.[27] Any missing CFCs would be the amount removed without affecting stratospheric ozone.[28]

Estimates of CFCs released into the atmosphere are available and are believed to be accurate to within five percent. However, scientists have had considerable difficulty in collecting data on tropospheric concentrations of CFCs. Some data have been collected for approximately ten years, but in 1978 the National Bureau of Standards studied the techniques used to measure tropospheric CFC concentrations and concluded that the measurements are subject to nonrandom calibration errors of ±40 percent.[29] The accuracy and comparability of measurements from different parts of the world are, therefore, very questionable. It should be noted that analyses of the currently existing data do not disprove the assumption of a near infinite tropospheric lifetime for CFCs. However, the uncertainty limits are quite large and range from no tropospheric removal of CFCs to a removal rate six times faster than that of stratospheric photolysis.

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[24] "Nonaerosal Propellant Uses ...," *op. cit.*, page 6.

[25] *Ibid.*, page 6.

[26] *Ibid.*, page 5.

[27] *Halocarbons, Effects On* ..., *op. cit.*, page 9.

[28] *Ibid.*, page 10.

[29] For the full National Bureau of Standards report see "Evaluation of Methodology for Analysis of Halocarbons in the Upper Atmosphere: Phase I," NBSIR 78-1480. For a summary of the NBS findings see, "Monitoring Halocarbons in the Atmosphere," by Dr. John Gribbon, *New Scientist*, January 18, 1979.

To resolve the uncertainty about tropospheric lifetimes, the Chemical Manufacturers Association (CMA) is supporting a worldwide program of highly accurate measurements of CFCs. Four measurement stations have been set up to monitor tropospheric CFC concentrations in Samoa, Barbados, Ireland, and Tasmania. Special care was taken to specially design the equipment to ensure comparability and accuracy of results. Reserves of test gas with known quantities of CFCs are being kept to periodically check the calibrations. Results from these monitoring stations could add significant new information to the body of knowledge concerning the propensity of CFCs to deplete stratospheric ozone.

C.2.3 Effects of Ozone Depletion

Increased UV-B

The primary concern over ozone depletion is from ultraviolet (UV) radiation of certain damaging wavelengths that will reach the earth's surface in increasing amounts. Exhibit C-5 shows the spectrum of light. The portion of the spectrum that is potentially harmful to plants, animals, and humans is that portion below 320 nm. Radiation of wavelengths below 180 nm does not reach the upper stratosphere; radiation from 180 nm to 240 nm does not reach the lower stratosphere because of screening by molecular oxygen; radiation from 240 nm to 290 nm does not reach the earth's surface due to screening by ozone; and some radiation from 290 nm to 320 nm, called UV-B radiation, reaches the earth's surface but the lower wavelengths are screened out by ozone. Ozone depletion will permit increased amounts of UV-B to reach the earth's surface, but the other wavelengths will not be affected. (As mentioned above, CFCs absorb radiation of wavelengths around 200 nm when they photodissociate in the upper atmosphere.)

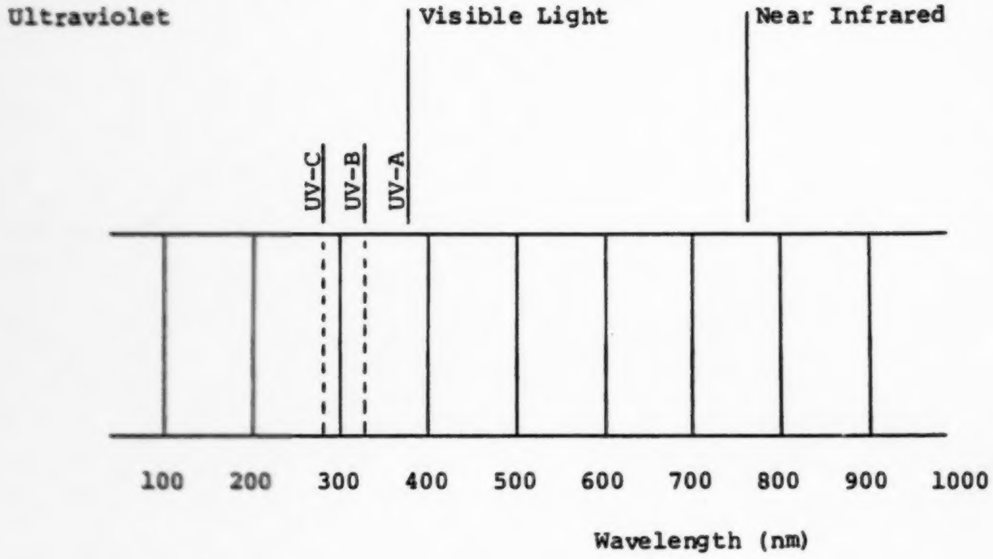
The shorter the wavelength of UV-B light, the greater its adverse biological effects. The relative effectiveness of various wavelengths of UV-B in causing skin cancers or other adverse biological effects (i.e., altering DNA) is roughly as follows:

320 nm -	0.03
315 nm -	0.1
310 nm -	0.6
305 nm -	2.6
300 nm -	15.
295 nm -	60.
290 nm -	160.

(Note: The maximum is 1000 at 265 nm.)

EXHIBIT C-5

THE SPECTRUM OF LIGHT



Ozone depletion causes the incidence of the various wavelengths of UV-B radiation at the earth's surface to be increased by different amounts. Therefore, a weighted average of the potential damage from increased UV-B has been developed and is called the "damaging UV" or DUV intensity.[30] It is the percent DUV increase that is of prime concern.[31] For the current range of predictions of ozone depletion the average percent increase in DUV per percent decrease in ozone is about 2.0 for the continental United States. This is referred to as the "physical amplification factor." Exhibit C-6 displays the change in DUV as a function of ozone depletion.

Biological (Non-Human) Effects

Currently, there is little or no information on the effect that increased DUV may have on plants and animals. Research is being conducted to determine the effects DUV may have on:

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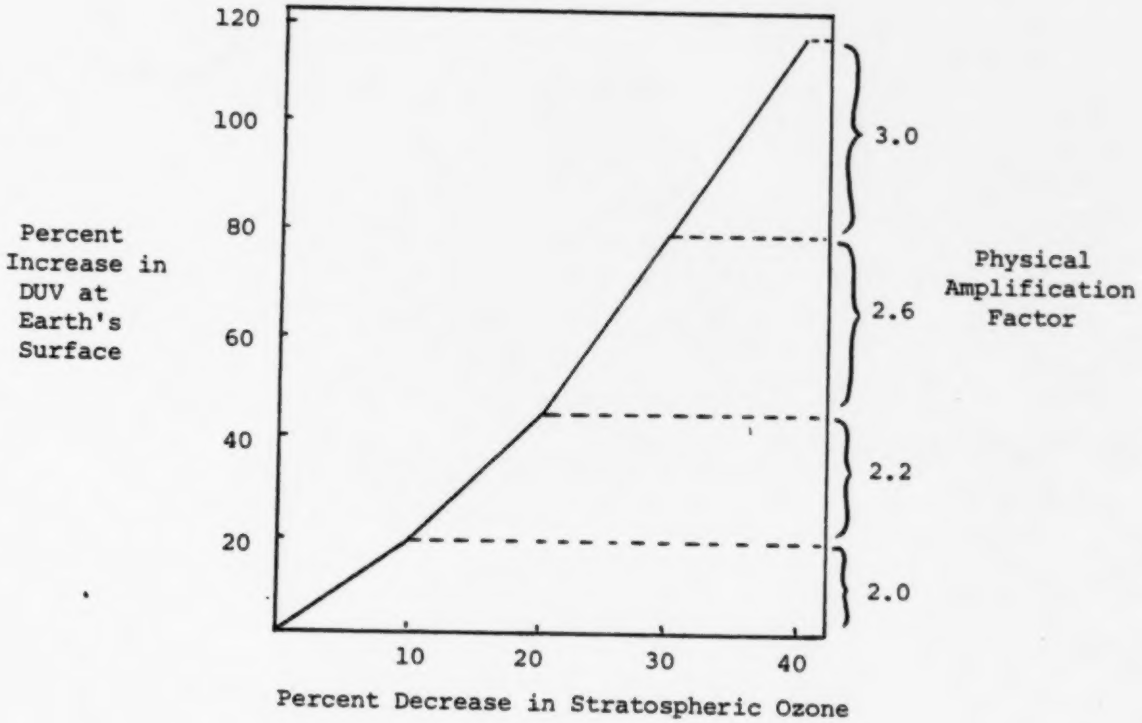
[30] Final ETS ..., op. cit.

[31] The corresponding increase in DUV is greater than the increase in UV-B because the increase is proportionately greater at shorter wavelengths.

EXHIBIT C-6

PHYSICAL AMPLIFICATION FACTOR



- species-poor ecosystems that might be very susceptible to alteration, for example at margins of deserts;
- the productivity of agricultural plants;
- the skin and eyes of terrestrial animals; and
- aquatic organisms near the surface of the water, particularly small organisms and eggs.

Existing information suggests that increased DUV can have adverse effects in all of these categories. However, it can not be proved or disproved that a sixteen percent decrease in stratospheric ozone would cause a significant reduction in yields from agricultural crops or non-agricultural land or aquatic ecosystems.[32] The uncertainty stems, in part, from ignorance of the proper weighting function for different wavelengths of UV-B to be used in comparing solar and artificial source irradiations that are used in tests.[33] More research is needed before the magnitude of these effects can be estimated.

Climate Effects

Ozone depletion upsets the natural heating and cooling capacity of the earth and its atmosphere. It is not possible at this time to predict what the net result will be.[34] Further study is proceeding in this area.

CFCs themselves can also cause climate changes. Current estimates are that at 1973 emission rates, CFCs could cause the average temperature in the lower atmosphere to rise by 0.5°C in the next 100 years. This effect is swamped by the predicted "greenhouse" effect of increased carbon dioxide (CO_2) abundance which is expected to produce an average global temperature increase of 0.5°C by the year 2000.[35]

Human Health Effects

Considerably more work has been done in the area of human health effects from increased exposure to DUV. This is not because it is believed that possible health effects will be more serious than the above mentioned effects on plants, animals, and the climate. However, whereas work has just begun in those areas, the effects of UV-B radiation on humans has been studied for a longer period of time, in no connection to the problem of ozone depletion. The primary human health effect from increased DUV is an increase in the incidence of both melanoma and nonmelanoma skin cancer.

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[32] Protection Against Depletion of Stratospheric Ozone by Chlorofluorocarbons, National Research Council, National Academy of Sciences, 1979.

[33] Ibid., page 69.

[34] Ibid., page 106.

[35] Ibid., pages 117-118.

Melanoma skin cancer (MSC) is a rare malignant cancer that is often fatal. (The age-adjusted five-year survival rate in recent years is about 67 percent.) The incidence of melanoma skin cancer is about one percent of the incidence of nonmelanoma skin cancer and is about 4/100,000 white population in the northern U.S. and 8/100,000 white population in the southern U.S. (See Exhibit C-7.)

Although at one time there was believed to be only a small correlation between DUV exposure and MSC, current evidence indicates that the development of malignant melanoma skin cancer in man is related to DUV exposure.[36] In its 1979 report on ozone depletion, the National Academy of Sciences stated that "the relation of solar UV radiation to melanoma (skin cancer) ought to be taken as a likely health hazard of a significant size and responded to accordingly."[37] Therefore, current estimates of the health effects induced by ozone depletion should include increases in the incidence of MSC.

The correlation between nonmelanoma skin cancer (NMSC) and exposure to UV-B is well documented. The logarithm of its incidence is inversely related to latitude and directly related to flux of UV-B. This cancer is much less serious than melanoma skin cancer and is almost never fatal. An NMSC victim can usually be treated as an outpatient at a direct cost of approximately $200 per visit (1978 dollars).

A considerable amount of work has been done to improve estimates of the dose-response relationships between MSC, NMSC and DUV. Although the relationship between MSC and DUV appears to be more complicated than the relationship between NMSC and DUV, they are of about the same general shape; a one percent increase in DUV exposure results in a two percent increase in the incidence of each of the skin cancers.[38] This is referred to as the "biological amplification factor." A biological amplification factor equal to 2.0 is consistent with the preliminary results of a National Cancer Institute survey of caucasians in the continental U.S. Incorporating the physical amplification factor, also equal to 2.0, a one percent reduction in stratospheric ozone yields a four percent increase in both NMSC and MSC:

1% Depletion of Ozone → 2% Increase in DUV → 4% Increase in MSC and NMSC .

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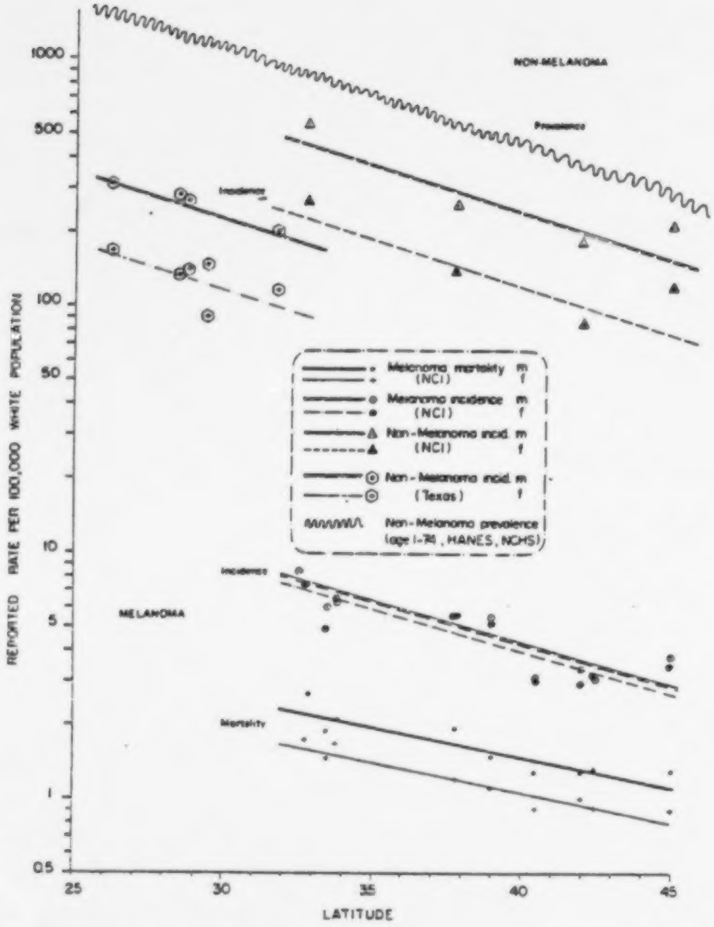
[36] *Ibid.*, page 74.

[37] *Ibid.*, page 85.

[38] *Ibid.*, pages 103.

EXHIBIT C-7

RELATIONSHIP BTWEEN MSC, NMSC, AND LATITUDE



Reported skin-cancer rates among whites as a function of latitude. [Sources: melanoma mortality from Mason and McKay (1974), melanoma incidence from National Cancer Institute (1974), nonmelanoma skin-cancer incidence (NCI) from Scotto *et al.* (1974), nonmelanoma skin-cancer incidence (Texas) from Macdonald (1974), and prevalence of nonmelanoma skin cancer based on preliminary data from the Health and Nutrition Examination Survey of the National Center for Health Statistics (McDowell, 1974).]

Source: Protection Against Depletion of Stratospheric Ozone by Chlorofluorocarbons, page 83.

There is, however, approximately a twenty year lag in the appearance of skin cancer caused by increased exposure to DUV. Therefore, the increases in incidence would not be seen immediately. Since ozone depletion also takes a very long time to reverse itself, once CFC emissions were reduced the increased incidence of skin cancer could be expected to result for many years.

To reduce the risk of contracting NMSC and MSC those individuals most susceptible could be identified and instructed to take precautions. Unfortunately, all light-skinned people are believed to be susceptible meaning that as much as one-third of the U.S. population would have to protect itself.[39] It has not been demonstrated that sunscreens (such as commercially sold suntan lotion) reduce the risk of either NMSC or MSC and most clothing transmits more than twenty percent of DUV. Also, clouds transmit seventy to eighty percent of DUV. Therefore, the most effective method to reduce the risk of skin cancer is for susceptible individuals to refrain from exposures between 11:00 a.m. and 3:00 p.m.[40] This would require a significant change in recreational and occupational lifestyles for most susceptible individuals.

C.2.4 Quantification of the Benefits of the Aerosol Ban

Magnitude of Environmental Effects

The environmental effects of CFC emissions are characterized by the effect that increased DUV radiation, due to ozone depletion, might have on humans, plants and animals, and the climate. The effect on plants and animals and the climate have not been estimated. Although some evidence suggests that increased DUV radiation could have a significant adverse effect on crop yields and the aquatic food chain, there is too much uncertainty to quantify these effects.[41] General speculation suggests that in the U.S., the costs of crop damage could be high because the value of agricultural products grown in the U.S. is very high.

Similarly, the costs and benefits of a CFC induced change in climate have not been estimated. Even the sign of a CFC induced temperature change cannot be determined with certainty. Therefore, any benefits (or costs) that result from reducing the effect that CFCs have on our climate cannot be quantified.

Magnitude of Human Health Effects

The effect on human health of increased DUV radiation is the only area in which sufficient work has been done to support quantification. The primary health effect, as described above is increased incidence of both melanoma and nonmelanoma skin cancers. A major issue that must be addressed when estimating the magnitude of these predicted health effects is whether future health effects, including deaths, should be discounted, and if so, at what rate. This question arises in virtually all environmental regulations and there are no clear-cut, universally accepted solutions, as discussed in Chapter 2.

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[39] Protection Against Depletion ..., op. cit., page 343.

[40] Ibid., page 343.

[41] Ibid., page 69.

For purposes of illustration, we will analyze the health benefits of the CFC regulation under two different discounting assumptions. As mentioned in Chapter 2, the appropriateness of a non-zero discount rate for health benefits is unclear. A policy decision by EPA is needed on this issue.

If health benefits are to be discounted, the outcome of a cost-effectiveness analysis on CFC use is very sensitive to the choice of a discount rate. The economic costs of banning aerosol uses of CFCs accrue immediately while the health benefits accrue primarily in the distant future. Varying the discount rate from zero percent to ten percent will change the present value of the health benefits by two orders of magnitude while the effect on the economic costs will be smaller because the costs are incurred sooner. Therefore, the higher the discount rate, the lower the cost-effectiveness of reducing CFC emissions from aerosol uses. Results for discount rates of zero and five percent are presented in this case study.

To estimate the benefits derived from the ban, the health effects per pound of CFC emitted must be calculated. This quantity can then be multiplied by the number of pounds of CFC emissions that the ban prevented. Numerous assumptions are required to approximate the health effects per pound of CFC emitted. As described above, many uncertainties remain in the ozone depletion theory. Therefore, a range of the estimated benefits will be presented.

In the range of emission reduction that the ban has caused, the marginal effects of CFC emissions are equal to the total average effects (i.e., the relationships between emissions, increased DUV radiation, and increased incidence of skin cancers are constant). Therefore, to calculate the average health effects per pound of CFCs, we have assumed that CFC emissions remain constant worldwide for thirty years at 1975 levels; that there is a twenty year lag in the appearance of skin cancer from the time of exposure to the critical amount of UV-B; that there is a ten year lag in the transport of CFCs to the upper stratosphere; and that the current NAS Report prediction of 16.5 percent steady state ozone depletion is approximately correct. Given these assumptions, we estimate the pattern of health effects over seventy years due to emission of CFCs over thirty years to be:

PATTERN OF HEALTH EFFECTS OVER SEVENTY YEARS DUE TO CFC EMISSIONS OVER THIRTY YEARS

Year	CFC Emissions 10^6 Pounds/Yr.	Ozone Depletion	Percent Increase in Skin Cancers	Number of Increased Cases of Skin Cancer/Yr
1970	1,900	0%	0%	0
1980	1,900	2.0%	0%	0
1990	1,900	4.5%	0%	0
2000	0	7.0%	8%	34,800
2010	0	7.5%	18%	88,300
2020	0	6.2%	28%	153,300
2030	0	5.5%	30%	175,700
2040	0	5%	24.8%	147,000

The estimate of emissions over thirty years is the generally recognized scenario of constant emissions at 1975 world levels. The estimates of ozone depletion are taken from the NASA report The Stratosphere: Present and Future. The percent increase in the incidence of skin cancers is estimated using the "1:2:4 rule" (discussed above) with a thirty year lag, and the increased numbers of skin cancers is derived from FDA estimates of skin cancer incidence over the next one hundred years. The results are:

- Total increase in skin cancers over seventy years equals 6.0 million. Total emissions of CFCs over 30 years equals 57 billion pounds. The increased number of skin cancers equals 100 per one million pounds of CFCs.
- When both CFC emissions and skin cancers are discounted at five percent, the increased number of skin cancers equals 16 per one million pounds of CFCs.

Since the NAS predicted 16.5 percent steady state ozone depletion with an uncertainty of plus or minus 67 percent, the health effects per one million pounds of CFCs are also very uncertain. Taking into account only the uncertainty reported by NAS, when both CFC emissions and skin cancers are discounted at five percent, the increased number of skin cancers may be as low as 5 per one million pounds of CFCs. Without discounting, the increased number of skin cancers may be as high as 170 per one million pounds of CFCs. Exhibit C-8 displays the full range of these results.

To translate these estimates into the benefit of the ban, the amount of emissions prevented by the ban must be approximated. In the early 1970s, CFC emissions from aerosol uses grew from approximately 350 million pounds per year to 450 million pounds per year. In 1974, when the ozone depletion controversy first began, the annual emissions from aerosol uses of CFCs began to decline.

EXHIBIT C-8

RANGE OF NUMBER OF CANCERS EXPECTED PER ONE MILLION POUNDS OF CFC EMISSIONS GIVEN DIFFERENT DISCOUNTING PROCEDURES AND DIFFERENT ASSUMPTIONS CONCERNING THE PROPENSITY OF CFCs TO DEPLETE OZONE

	"Low" Case[1]	"Medium" Case[2]	"High" Case[3]
No discounting.	30	100	170
Both number of cancers and pounds of CFC emissions discounted at five percent.	5	16	27

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[1] "Low" Case = NAS prediction of 5 percent steady state ozone depletion given constant emissions at 1975 rates.

[2] "Medium Case = NAS prediction of 16.5 percent steady state ozone depletion given constant emissions at 1975 rates.

[3] "High" Case = NAS prediction of 28 percent steady state ozone depletion given constant emissions at 1975 rates.

By 1977 when the notice of the proposed CFC aerosol propellant ban was first published, the annual use of CFCs as aerosol propellants dropped to 175 million pounds. The amount of CFC emissions actually prevented by this rule is uncertain because there is no way to know exactly what would have happened in the absence of the rule.

It is unlikely that aerosol uses of CFCs would have jumped back to their high of the mid-1970s. Capital conversions had already begun in the aerosol industry and it probably would not have been worthwhile for industry to switch back. Most, if not all, of these changes were made in anticipation of the rule, however, and in the scenario where the rule did not exist it may be appropriate to assume that the anticipation of the rule also did not exist. Therefore, one can reasonably construct at least two levels of CFC aerosol emissions prevented by the rule. We shall construct two such scenarios.

For the first scenario, Scenario 1, we will assume that CFC emissions from aerosol propellant uses would have continued to decline at its post 1974 rate. For the second scenario, Scenario 2, we will assume that aerosol propellant uses of CFCs would have continued at their 1976 level for five years and then slowly tapered off over the next ten years. This tapering off is justified because less controversial substitute propellants would likely be developed during that time. Exhibit C-9 displays these two scenarios.

Under Scenario 1, total emissions prevented are 175 million pounds. Under Scenario 2, the total emissions prevented are 2,300 million pounds. If no discounting is used, and the NAS "medium" prediction of 16.5 percent steady state ozone depletion is assumed, the total number of cancers prevented for each scenario is:

- Scenario 1: (100 cancers per 10^6 lbs) x (175 x 10^6 lbs) = 17,500 cancers prevented.
- Scenario 2: (100 cancers per 10^6 lbs) x (2,300 x 10^6 lbs) = 230,000 cancers pervented.

In the absence of the regulation, these cancers would have occurred over a period of approximately ten years, beginning thirty years from now (or approximately in the year 2010). Of these cancers, one percent are expected to be melanoma skin cancers which result in premature death approximately one third of the time. Therefore, the number of premature fatalities prevented by the aerosol propellant ban under these scenarios for the above assumptions are:

- Scenario 1: (17,500 cancers prevented) x (1 case of MSC per 100 cases of skin cancer) x (1 death per 3 cases of MSC) = 58 premature deaths prevented.
- Scenario 2: (230,000 cancers prevented) x (1 case of MSC per 100 cases of skin cancer) x (1 death per 3 cases of MSC) = 770 premature deaths prevented.

EXHIBIT C-9

ESTIMATES OF CFC EMISSIONS PREVENTED BY 1978 CFC AEROSOL PROPELLANT BAN

		Scenario 1[1]		Scenario 2[1]	
Year	Actual CFC Aerosol Emissions[2]	Predicted CFC Aerosol Emissions[2]	Emissions Prevented[2]	Predicted CFC Aerosol Emissions[2]	Emission Prevented[2]
1970	340	340	-	340	-
1971	350	350	-	350	-
1972	390	390	-	390	-
1973	460	460	-	460	-
1974	440	440	-	440	-
1975	360	360	-	360	-
1976	300	300	-	300	-
1977	170	170	-	300	130
1978	80	100	20	300	220
1979	0	60	60	300	300
1980	0	40	40	300	300
1981	-	25	25	270	270
1982	-	15	15	240	240
1983	-	10	10	210	210
1984	-	5	5	180	180
1985	-	0	-	150	150
1986	-	0	-	120	120
1987	-	0	-	90	90
1988	-	0	-	60	60
1989	-	0	-	30	30
1990	-	0	-	0	-
			175		2,300

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[1] Scenario 1: Aerosol propellant uses continue to decline at post 1974 rates event without the ban.

Scenario 2: Aerosol propellant uses remain stable at 1976 levels for five years and then decline slowly over ten years.

[2] Millions of pounds.

Exhibit C-10 shows the full range of prevented premature deaths and cases of nonfatal MSC that can be calculated taking into account the uncertainty of the NAS ozone-depletion estimate and different approaches toward valuing future effects (i.e., discounting).

The validity of the estimates shown in Exhibit C-10 is easily questioned because the entire analysis is based on a hypothesis. The ozone-depletion theory has not been proved, and many uncertainties remain. However, given the best current knowledge available (as represented by 1979 National Academy of Sciences and NASA publications), and given various assumptions about valuing and predicting future behavior and events, the number of premature deaths in the United States prevented by the CFC aerosol propellant ban is very likely to be in the range of 58 to 770. These predictions do not include:

- effects on people in other countries;
- effects on the environment;
- increases in the number of untreated cases of nonmelanoma skin cancer that result in premature death;
- potential improvements in the effectiveness of MSC treatments;
- changes in lifestyles that significantly affect exposure to the sun; and
- changes in the susceptibility of the population to melanoma and nonmelanoma skin cancers.

EXHIBIT C-10

ESTIMATED PREMATURE DEATHS AND NONFATAL MELANOMA SKIN CANCERS PREVENTED BY THE CFC AEROSOL PROPELLANT BAN FOR VARIOUS ASSUMPTIONS[3]

	Scenario 1[1]			Scenario 2[1]		
	NAS "Low" Case[2]	NAS "Medium" Case[2]	NAS "High" Case[2]	NAS "Low" Case[2]	NAS "Medium" Case[2]	NAS "High" Case[2]
No Discounting	17/35	58/115	100/200	230/460	770/1540	1300/2600
Both Cancers and Pounds of CFC Emissions Discounted at Five Percent	3/6	9/18	16/32	38/76	120/240	200/400

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[1] Scenario 1 = 175 million pounds of CFC emissions prevented by the ban.

Scenario 2 = 2,300 million pounds of CFC emissions prevented by the ban.

[2] "Low" Case = NAS prediction of 5 percent ozone depletion.
"Medium" Case = NAS prediction of 16.5 percent ozone depletion.
"High" Case = NAS prediction of 28 percent ozone depletion.

[3] Premature Death/Nonfatal Case of Melanoma Skin Cancer.

C.3 COSTS AND ECONOMIC EFFECTS OF THE CHLOROFLUOROCARBON BAN

C.3.1 EPA Estimates of Economic Impact

The prospective economic impact of the ban on CFC's was evaluated by International Research and Technology under contract to EPA.[42] Nine regulatory scenarios were evaluated which included all combinations of three phase-out periods (30 months, 18 months, and 6 months) and three allowable substitution levels (limited fluorocarbon ban, complete fluorocarbon ban, complete aerosol ban). The final rule ultimately promulgated on March 17, 1978 provided for a nine-month phase-out period and banned only fully halogenated CFCs (not other flurocarbons). However, positive mutagenicity tests on other fluorocarbons has created uncertainty about the substitutability of other fluorocarbons for those banned. As of 18 months ago, other fluorocarbons were not being used as substitutes. In addition, the selection of an appropriate phase-out period for purposes of estimating economic impact is not clear. Because it can be documented that affected firms began modifying their behavior at the time of the proposal (May 1977), the eighteen month scenario might be more appropriate. Therefore, it will be assumed that actual economic effects can be best approximated by the regulatory scenario represented by a complete fluorocarbon ban and an 18 month phase-out period.

Methodologically, the economic impact of a ban raises completely different issues from estimating economic impacts of an engineering control regulatory intervention. In the engineering control case, direct capital and operating costs are incurred by suppliers in well-defined markets. In essence, the regulation shifts the supply curve in these directly-affected markets because there are now increased capital and operating costs associated with producing the same quantity of goods. Economic effects flow from these direct costs depending upon how suppliers react to the increased costs. Choices facing suppliers include closing down or continuing operations with some pass-through of increased costs due to compliance.

In the case of a ban, the supply curve does not just undergo a shift in the directly affected markets -- it is eliminated. Such a major perturbation must have major effects up and down the relevant distribution chains and in the markets for compliments and substitutes as well. There is no data source comparable to an engineering cost estimate for some model plant from which to estimate the direct cost impact. Instead, the economic analysis must rely on perceived demand shifts and how they are met. The economic impact is then estimated by the loss in value added when moving from a baseline demand scenario to an appropriate regulatory demand scenario.

Value added is the difference between the cost of input materials and the revenue received from output goods and is essentially the contribution of capital, labor, and energy. Each stage of the distribution chain contributes to value added, so the differences in value added need to be considered over the entire distribution chain (and possibly over the distribution chain of alternative products as well). In general, bans will not be promulgated unless it is perceived that there is some adequate substitute, so the major economic effects of bans lie in the shocks to existing distribution chains rather than in some sort of major reduced consumer surplus. The concept of value added already implicitly includes such secondary economic effects as employment and prices so it would be incorrect to count such effects in addition to value added. However, if labor intensiveness and price elasticities are known, these secondary economic effects can be isolated from the overall estimate of reduction in value added.

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[42] The Economic Impact of Potential Regulation of Chlorofluorocarbon - Propelled Aerosols, IR&T, April 1977, prepared under EPA contract 68-01-1918.

In the chlorofluorocarbons case, the approach was as follows:

- define the relevant universe as distribution chains for aerosol products;
- develop appropriate demand scenarios based on expected consumer behavior;
- develop cost, price and production data for each stage of the distribution chain (this part was not well documented, but it will be assumed that it was based on the best available data); and
- estimate economic impact in terms of reduced value added based on changes in demand.

For the aerosol industry (chemical manufacturers, valve manufacturers, container manufacturers, fillers, and marketers), reduction in value added for the four years 1977-1980 (calculated in millions of 1976 dollars) was estimated as follows:

Marketers	$ 573	
Fillers	26	(including offset for filling hydrocarbon and mechanical systems)
Container Manufacturers	132	
Value Manufacturers	37	
Chemical Manufacturers	299	
Total 4 Year Impact	$1,067	

These impacts were developed under the assumption that no segment attempts to recover its loss through price increases. Should price increases occur, there would be a redistribution of the four-year impact. Because aerosol products would compete with mechanical spray products (which are apparently cheaper than aerosols), it is not likely significant price increases would occur.

C.3.2 Adjustments to EPA Estimates

There are three issues which need to be addressed in adjusting the EPA estimates of reduction in value added:

- coverage of the analysis was limited to the aerosol industry;
- appropriate time-horizon was limited to four years; and
- economic impact assessment for the chemical industry is not consistent within the report

These are addressed below.

Coverage

Limiting the analysis to the aerosol industry necessarily focuses on those industry segments who will be net losers due to the ban. In reality, the losses to the aerosol industry will be offset by gains in the mechanical filling industry (plastic and glass containers, mechanical pumps, etc.). No attempt was made to account for these offsetting effects in the EPA analysis. These offsetting effects would tend to make the EPA analysis overstate the actual overall economic impact.

Time Horizon

No justification is given for the selection of the four-year period. Although it certainly seems reasonable that economic effects attributable to this ban do not go on infinitely, it may be reasonable for the attributable effects to last more than four years. Although this factor works in the opposite direction from the coverage factor, it would not be of the same magnitude. In general, economic effects should probably not be measured past the point at which firms have exited from the aerosol industry due to the ban, suppliers have converted their plants, and marketers have reformulated their products. At its maximum, this period should not be much more than a year or two beyond the assumed four years.

Internal Inconsistency

The EPA analysis contains an internal inconsistency regarding the impact of the chemical producing segment. In the Executive Summary, the total impact on chemical manufacturers was estimated at $299 million (pg. xviii) yet in the analysis itself, the economic impact to this segment is estimated as $118 million (pg 4-46). No explanation is given for why the impact to this segment increased by $181 million from the analysis to the Executive Summary.

Assuming the $229 million to be in error, the average annual impact of the ban needs to be reduced by $181 million. In addition, there needs to be some reduction in the overall economic impact in order to account for the coverage underestimate. Based on preliminary data regarding products available in 1980 and their prices[43] (broad range of products available at prices consistent with general inflation increases), it does not seem likely that the CFC ban led to a significant reduction in consumer surplus. Therefore, it is likely that most losses to the aerosol industry have been offset by gains to the mechanical filling industry. Data on recent changes in the mechanical filling industry are required to assess the extent to which gains in this industry actually offset losses in the aerosol industry. In the absence of such data, it may be reasonable to reduce the overall economic effect by even as much as 25 percent and still have it remain conservative. Whether or not annual economic effects should be counted for more than four years is a matter of conjecture, but it certainly should not be counted more than one or two additional years. The adjustments are as follows:

Total 4-year impact as presented in analysis	$1,067 million (1976 $)
Reduction for chemical industry inconsistency	$ 886 million (1976 $)
Average annual impact	$ 222 million (1976 $)
Average annual impact in 1979 dollars	$ 275 million (1979 $)
Average annual impact reduced by 25 percent to adjust for coverage	$ 206 million (1979 $)

C.3.3 Secondary Economic Effects

The major secondary economic effects are potential changes in market structure due to the large number of small businesses, particularly in the filling segment. According to the EPA report, marketers, container manufacturers, valve suppliers and chemical producers are significantly large to be able to finance any necessary adaptations. However, there are many small fillers who process a limited number of aerosol products for the bulk of their business. Shifting from a fluorocarbon system to a hydrocarbon system will be very difficult for these firms.

The EPA analysis also attempted to estimate employment losses attributable to the ban. By isolating the part of value added attributable to labor at each stage, the dollar value of labor lost was estimated. Using a direct labor cost of $10,400 per man-year (1976 dollars), the net loss was estimated at 2,000 jobs or $83 million for the four year period. Of course, this estimate needs to be balanced against job increases in the mechanical filling industry. Again, data on the mechanical filling industry would be required. These employment estimates include direct labor only and do not include any indirect or administrative job loss.

C.3.4 Data on Actual Economic Effects

A retrospective analysis of the chlorofluorocarbon ban is currently being conducted by JACA Corporation for EPA. Preliminary data presented in their methodology paper indicates an upturn for metal aerosol can output after

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[43] "Methodology for Retrospective Analysis of the Economic Impact of the U.S. Government Ban on Chlorofluorocarbons for Aerosol Uses," JACA Corporation, July 1980.

1977 and a continual decline in CFC aerosol use. This would indicate an increase in hydrocarbon systems, consistent with the assumed regulatory scenario. Additional data from this retrospective study should be available shortly.

C.4 COST EFFECTIVENESS ANALYSIS

The range of benefits developed in Section C.2 and the costs reported in Section C.3 can be used to estimate a range of cost-effectiveness for the CFC aerosol ban. For the case when no discounting is used, the number of premature deaths prevented are likely to be between 58 and 770 and may range from as low as 17 to a high of 1,300. In addition, twice as many nonfatal cases of melanoma skin cancer and 100 times as many cases of nonmelanoma skin cancer are predicted to have been prevented. The upper bound on total cost is reported as $1.1 billion and may be adjusted as described in Section C.3 to approximately $825 million to account for inadequate coverage of industries that benefit from the rule (1979 dollars). Exhibit C-11 displays the full range of cost-effectiveness of the regulation that may result from these predictions and estimates.

Since the different types of benefits cannot be combined into a single measure, Exhibit C-11 was constructed by dividing the total estimated cost by each of the predicted benefits. This clearly overestimates the cost for obtaining each different type of benefit. However, any division of the costs among the different benefits would be arbitrary because one benefit cannot be obtained without the others. One way to solve this problem would be to use reduced future medical costs as a lower bound of the value of preventing cases of nonfatal MSC and NMSC. This estimate of those two benefits could be subtracted from the total cost and the remaining cost estimate could be used to develop a cost-effectiveness measure per premature death prevented.

Exhibit C-11 displays the sensitivity of the cost-effectiveness measures to assumptions concerning both the propensity of CFCs to deplete ozone and the number of pounds of CFC emissions prevented by the ban. The cost of preventing a premature death from MSC is most likely in the range of $1.5 to $19 million in 1979 dollars. (For a five percent discount rate, the numbers in Exhibit C-11 should be multiplied by approximately six.) With information on the age distribution at which MSC is contracted, an estimate of the cost per "adjusted life-year saved" could be developed. Cost-effectiveness estimates in terms of cost per "adjusted life-year saved" could be compared to similarly constructed cost-effectiveness measures for other regulations.

Benefits that are not included in the analysis are:

- Adverse effects on the natural environment prevented;
- Adverse effects on agriculture and domestic livestock prevented; and
- Health effects on people in other countries prevented.

EXHIBIT C-11

RANGE OF COST-EFFECTIVENESS IN MILLIONS OF 1979 DOLLARS FOR THE CFC AEROSOL PROPELLANT BAN WHEN DISCOUNTING IS NOT USED[1]

	"Low" Case[2]	"Medium" Case[2]	"High" Case[2]
Cost per premature death prevented.[3]	5 to 65	1.5 to 19	0.8 to 11
Costs per nonfatal case of MSC prevented.[3]	2.5 to 33	0.8 to 10	0.4 to 6
Cost per case of MSC prevented.[3]	0.05 to 0.6	0.01 to 0.2	0.01 to 0.11

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[1] Range shown is for two scenarios:

Scenario 1 = 175 million pounds of CFC emissions prevented by the ban;
Scenario 2 = 2,300 million pounds of CFC emissions prevented by the ban.

[2] "Low" Case = NAS prediction of 5 percent ozone depletion.
"Medium" Case = NAS prediction of 16.5 percent ozone depletion.
"High" Case = NAS prediction of 28 percent ozone depletion.

[3] Total cost of the regulation is taken as $1,100 million. All estimates could be reduced 25 percent to compensate, in part, for inadequate coverage of industries expected to benefit from the ban.

The benefits of preventing people in other countries from contracting skin cancers may be of the same magnitude as the benefits estimated for the U.S. It is not possible to assess the magnitude of the other benefits.

As discussed in Section C.3, the cost estimates are likely to be upper bounds since they may not fully account for the industries that benefit from the ban. However, they also do not include the costs imposed by the hazards of using hydrocarbons as substitutes for CFC propellants. Hydrocarbons are flammable and their use may increase the number of burns or the amount of property loss due to fires. It is not possible to assess the magnitude of these costs at this time.